The potential of restored landfill sites to support pollinating insects

Submitted for the Degree of Doctor of Philosophy
at the University of Northampton

2009

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Abstract

Habitat restoration is an important tool in reducing the current decline in biodiversity. To determine the success of restoration, ecologists have previously focused on species richness or on the presence of rare species; little is known of species interactions. This study examines both the potential of restored landfill sites to support pollinating insects and how flower-insect interactions can be used in determining successful restoration. These are important attributes of ecosystem function. Standard belt transects were used to record flowering insect pollinated plants and flower-visiting insects on nine paired restored landfill and reference nature sites, in the broader Northamptonshire region (UK). Over the duration of this study, an area of 25,000m^2 was surveyed for floral characteristics and approximately 138,000 floral units were counted from 98 plant species. A total of 201 flower visitor surveys were performed, with 942 flower-visiting insect samples taken. Flowering plant species richness and abundance of floral resources on restored landfill sites were not found to be significantly different from those on reference sites and the flower-visiting insect assemblages were similar in terms of species-richness and abundance. Interaction structures were examined and whilst the plant-insect assemblages had few species in common, both showed similar levels of nestedness and connectance. The differences in the species but similarity in the functioning of these assemblages emphasise the importance of examining interaction structures within a functional approach to the evaluation of restoration. There are 2,200 landfill sites in England and Wales covering some 28,000 ha, and this study highlights that their restoration can potentially provide an important resource for the conservation of pollinating insects and the services that they provide for both natural and agricultural plants.

Acknowledgments

This project has been generously supported by the SITA Trust from the Landfill Communities Fund, for which I am very grateful. Without the help of many people I would not have been able to start or complete this project. Firstly I would like to thank my great supervisors Jeff Ollerton and Duncan McCollin, whose guidance and help has been immeasurable, both whom have invested their time and energy into me and I hope I can someday repay their faith. Secondly I would like to thank Lutfor Rahman and Joanna Griffin who given much support and have kept me sane. Thanks to the landfill site operators and personnel who have made this research possible:

Sean Matty – Sita SITA

Nigel Rogers – Barton Plant Limited

Nick Eldridge / Colin Richardson / Helen Jackson / Katie Burfitt - WRG

Timothy Bigelow – Viridor

Ed Burnett – Bedford CC

Nick Tipping & Rob Jones- Biffa

For the reference nature site managers who are leading the way in nature conservation:

Chris Coppock – MK Council

Ian Wilson and The Wildlife Trust BCNP

Nicola Orchard – Natural England

Eddie Asbrey and The Warwickshire Wildlife Trust

Special thanks also go to:

Professor Ian Livingstone and Dr. Jane Stout

Dr. Chris Holt and Professor Victor Parra-Tabla

Dr. Janet Jackson and Dr. Margaret Bates

Tina Fairless- a fantastic technician.

John Showers - Northamptonshire Hoverfly group

Field assistants: Lee Muttock & Claire Heptinstall

Many other people who have contributed to such an enjoyable time spent at the University of Northampton and whilst at SCAPE meetings for such a great introduction to how conferences should be.

For my family and friends, I thank you for your motivation, inspiration and love.

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Chapter descriptions, aims and objectives

The overall aim of this project was to explore the potential of restored landfill sites to support pollinating insects, to determine the floral resources, the pollinator richness and abundance and the interaction structure. A brief summary of the research questions are included in the following chapter descriptions:

Chapter 1 – Introduction

An overview of themes in the literature relevant to the project. Within the introduction chapter, the following ideas will be developed.

- **Habitat loss and the importance of restoration**
    - Habitat restoration methods
        - Grassland restoration and seed mixes vs. natural colonisation
        - Post-creation remediation methods
    - Measuring the success of restoration
        - Environmental interactions
- **Pollination and pollinating insects**
    - Flowering plants and Floral resource provision
    - Interaction structure of plants and flower-visiting insects
    - Pollinator decline and conservation
    - Scales of effect on pollinating insects
        - Habitat quality
        - Landscape context
    - Habitat Quality *and* Landscape Context
- **Pollination on restored habitats.**
    - The Waste industry and Landfill restoration
    - Plant assemblages and pollinators on restored landfill sites.

Chapter 2 – General Methods

General methods: those methods which are used through out the thesis, critique of them, and alternatives available.

Chapter 3 - The floristic characteristics of restored landfill sites

The following research questions aim to determine this:

1. How does the richness of plants in flower and floral abundance compare between restored landfill sites and reference nature sites? How does the richness of plants in flower and floral abundance compare between landfill sites which have been sown and those which have naturally re-colonised?

2. What is the effect of soil characteristics on richness of plants in flower on restored landfill sites?

3. How do restored landfill sites compare with the reference sites for the provision of resources for flower-visiting insects?

Chapter 4 – A comparison of flower-visiting insects on restored landfill and reference sites.

The following research questions aim to achieve this:

1. How does the flower visitor abundance and species richness compare between restored landfill and reference nature sites?

2. How does floral resource use compare between landfill and reference sites?

Chapter 5 - The flower-insect interaction web on restored landfill sites.

The following research question aims to achieve this :

How does the flower-insect interaction structure for the restored landfill sites compare to the reference sites in relation to connectance and nestedness?

Chapter 6 – Conclusion, recommendations for landfill site operators and areas for further research

Can these results be used to understand the broader factors relating to the potential of restored landfill sites to support pollinating insects?

What recommendations can be made for landfill site operators with regards to restoration practice?

Areas for further research

Chapter

1

Introduction

The potential of restored landfill sites to support assemblages of pollinating insects

"No more bees, no more pollination, no more plants, no more animals, no more man."
Albert Einstein (1879-1955) (attributed)

Within this chapter, the following ideas and relevant research will be developed.

- **Pollinator decline and the global biodiversity crisis**
- **The importance of restoration**
- **Landfill restoration and the waste industry**
- **Pollination on restored sites**
- **General aims**
- **Chapter descriptions**

Further details relating to specific chapter areas are contained within the stand-alone research chapters 3-6.

Pollinator decline and the global biodiversity crisis

Populations of plants and pollinators are being lost due to habitat degradation and fragmentation across a number of regions of the world (Allen-Wardell et al., 1998; Corbet, 2000; Steffan-Dewenter et al., 2002). Possible reasons for these declines are many-fold and include factors such as: introduced species, disease, and climate change; but the biggest impact has come from the intensification of agriculture over the last 50 years. Agricultural intensification has had the effect of habitat loss and fragmentation and the increased use of agro-chemicals, including insecticides, herbicides and fertilisers. There has also been the development of monoculture crop landscapes, overgrazing and land clearance, which removes nesting sites and the wild flowers that provide diverse food resources whether the agricultural crops are in bloom or not.

The global human population has grown exponentially, from 791 million in 1750 AD to 6500 million in 2005 (U.N., 2006). This rapid increase in human population resulted primarily from better health services and the growing abundance of food. Agricultural land use covers more than 45% of the European Union (Henle et al., 2008), and the UK is now one of the most densely populated and intensively farmed countries in Europe, following its drive for self-sufficiency following WWII (Green, 1989). Between 1930 and 1984, 97% of unimproved grasslands in England and Wales were lost, and it is unlikely that this trend will have reversed (Dryden, 1997). A great deal of marginal agricultural land has been brought into intensive arable production; between 1947 and 1980, in England and Wales 110,000 miles of hedgerow and some 25% of semi-natural vegetation was removed (Countryside Commission, 1986; Green, 1989).

Many pollinating insect species are experiencing steep declines in population size (Kearns et al., 1998). For example, declines have been reported for bees in North West Europe (Williams, 1982; Williams, 2005; Kosior et al., 2007) and bumblebee declines in North America (Colla and Packer, 2008; Grixti et al., 2009). Declines in non-bee taxa include butterflies in central Europe (Wenzel et al., 2006), and hoverflies in North West Europe (Biesmeijer et al., 2006).

Pollinator conservation concerns have been raised in numerous publications over the last couple of decades (Williams, 1995; Buchmann and Nabhan, 1996; Withgott, 1999; Carvell, 2002; Goulson, 2003b; Steffan-Dewenter, 2003; Pauw, 2007). Much of the current crisis relates to the decline of pollination services provided by the managed European honey bee colonies and much less information is available concerning wild pollinators (Ghazoul, 2005). However, the available evidence suggests that many wild pollinators have declined dramatically in recent decades, in the UK, in Europe and globally (Williams, 1982; Buchmann and Nabhan, 1996; Westrich, 1996; Kearns and Inouye, 1997; Corbet, 2000; Biesmeijer et al., 2006).

There are various causes believed to be driving the decline. Bumblebees for example, have been decreasing in both diversity and abundance, mainly due to loss of habitat from agricultural intensification (Forup and Memmott, 2005a). Other drivers for declines in certain species and groups include those with specialised diets (Goulson et al., 2005), those species which are late in emerging (Fitzpatrick et al., 2007) and those which are climate specialists (Williams et al., 2007; Williams and Osborne, 2009).The research community seems to be in consensus that the decline in pollinators is an important issue in conservation biology (e.g. Allen-Wardell et al., 1998; Kearns et al., 1998; Kevan et al., 2002; Carvell et al., 2006; Morandin et al., 2007; Ricketts et al., 2008). Concerns about the worldwide decline in pollinator species has been acknowledged by the high profile Convention on Biological Diversity which launched the International Pollinator Initiative in 2002. This is an important issue, because pollinators provide an important ecosystem service for crops.

Ecosystem services, including climate regulation, water purification, spoil production, pest control and crop pollination, are critical to human survival (UNEP, 2005). Natural ecosystems and the human environment rely on interactions with animals and organisms. Such interactions perform a vast number of functions, including soil aeration, recycling nutrients, seed dispersal, nitrogen cycling, carbon sequestration, natural selection and providing pollination services.

Pollination is a vital process in almost all terrestrial ecosystems (UNEP, 2005). Pollination is the transfer of a pollen grain (containing the male gamete) from a flower's anther to the stigma of the same or different flower. Plants are sessile and so sexual reproduction in plants requires pollen vectors. Such cross pollination provides a movement of genes between plants. These vectors can either be abiotic or biotic. Abiotic modes of pollination include pollen movement via wind, water or gravity, whereas biotic modes of pollination are through the "use" of animals to assist the pollen transfer. Globally, pollinating animals include both vertebrate and invertebrates; vertebrates include birds, bats, lizards, and invertebrates include predominantly insects but also some arthropods (Carter, 1892; Kevan and Baker, 1983; Ollerton, 1999). The most diverse and numerous are the insect pollinators (Buchmann and Nabhan, 1996).

Insects comprise more than half of all living species (Strong et al., 1984). They are large, in terms of numbers, biomass, species diversity and functional roles, and play a crucial part in most ecosystems, creating and maintaining habitats for numerous other species (Hook, 1997). It is estimated that of the described metazoan species, approximately 80% are insects (Samways, 1992). Plants gain from mutualistic interactions such as pollination, herbivory protection and seed dispersal provided by insects (Burd, 1994). Insect species are succumbing to anthropogenic effects such as habitat loss and chemical pollutants; however getting accurate figures for their general decline is difficult, since it is estimated that only 5% globally of insect species have been described (Samways, 1992; Hook, 1997), hence some species may be disappearing before they are even identified. It is estimated that less than two thirds of the worlds' bee species have been identified, and even within highly studied areas such as Western Europe, estimates vary between 2000 – 4500 species (Williams, 1995; Buchmann and Nabhan, 1996).

Pollinating insect orders include such organisms as bees, wasps, ants (Hymenoptera), true flies (Diptera), moths and butterflies (Lepidoptera) and beetles (Coleoptera), and are vital for effective pollination in numerous wild plants and crops (Rothrock, 1867; Todd, 1879; Trelease, 1881). In terms of biodiversity conservation the pollinators play a pivotal role and their loss can have a cascading impact on other species (Bond, 1994; Allen-Wardell et al., 1998; Colla and Packer, 2008). Globally, it is estimated that approximately 85% of flowering plants use insects and other animal pollinators in their reproduction; this varies from about 76% in the temperate regions of the world to over 94% in the tropics (Ollerton et al., in prep.) and in Western Europe about 80% of the plant species are insect pollinated (Kwak, 1994). Pollinators perform an important role in ecosystems helping to maintain diversity and populations of wild plants and agricultural crops (Kearns et al., 1998; Kremen et al., 2004; Tscharntke et al., 2005). From a human viewpoint we should be interested in pollinators owing to thirty five percent of the global production volumes of food crops being dependent on pollinators (Klein et al., 2007). However, the extent of our dependence on pollinators has been questioned, since most obligate animal-pollinated crops are economically small scale (Ghazoul, 2005), and few major crop species depend exclusively or solely on animal pollination. Therefore in the UK in relation to crops the current loss of pollinators is not critical for human food supply.

Few natural areas are managed or valued for the services they provide, although many are managed for the goods they produce. This may be because the ecology of ecosystem services such as crop pollination is poorly known, limiting our ability to understand their value. Public perception is that bees and particularly honey bees are our primary pollinating insects. However, relying solely on honey bees will not be as successful as a community of native insect pollinators (Goulson, 2003b; Morandin and Winston, 2006; Winfree et al., 2007). Native unmanaged bee populations provide important pollination services to various crops (Kremen et al., 2004), and are generally more diverse and abundant near to natural habitat (Dormann et al., 2007; Klein et al., 2007). The best economical agriculture option is to provide habitat for native pollinators since they produce higher yields but require little land (Kremen et al., 2004).

Placing a value on the services of ecological systems is difficult since one cannot place a value on intangibles. However they are indicative of the importance and relative magnitude of ecological service provision. The value per year for global pollination services, based on research of 2005 values, was US$214 Billion (Gallai et al., 2009). There are a number of caveats relating to this figure such as relative value of crops, extent of pollen limitation and inflation, but it does give an indication of the immense value of pollination services.

Globally pollinators are suffering from the loss of suitable habitats (Kremen and Ricketts, 2000). In the UK, notable loss of pollinator habitats include the loss of unimproved flower-rich grasslands (formerly valued as pasture and for hay production) and the removal of hedgerows (Goulson, 2003b; Goulson et al., 2005; Carvell et al., 2006; Williams, 2006). The loss of habitats and semi-natural vegetation causes loss of food resources, suitable nesting sites and materials (Öckinger and Smith, 2007). Pollinating insects may depend on many life history factors being met in relation to their occurrence on a habitat site and their population numbers. These are not simply the presence of food resources but also nesting sites and materials; for example bees may need bare earth or holes in wood to nest in or use for building materials, and butterflies and hoverflies require alternative larval food sources (Table 1.01).

Table 1.01 Habitat requirements for principle pollinating insects (Sheperd et al., 2003).

Insect Order		Food source		Nesting
		Larvae	Adult	
Hymenoptera	Bees and wasps	Pollen	nectar / pollen	site and material
Lepidoptera	Butterflies and moths	plant host	nectar	plant host for eggs
Diptera	Flies	carrion / prey	nectar	n/a
Coleoptera	Beetles	Plant host / pollen	nectar / pollen	n/a

The typical generalist nature of many pollinating insects makes no single plant a determining factor for their persistence in a habitat. Populations of solitary bees have been found to be limited by nesting opportunities and can be enhanced by placing suitable nesting houses in their habitats (Tscharntke et al., 1998). Therefore, nesting sites are more often limiting factors than pollen and nectar resources (Gathmann and Tscharntke, 2002). Local variability of vegetation height has been found to affect pollinator composition on grassland sites in Sweden (Sjödin et al., 2008); here it was deduced that the increased biomass provided more food for herbivorous adult beetles and their larvae, whereas bee community composition was mainly determined by flower abundance.

At the local scale, insect species richness is often positively correlated with habitat quality. This is often measured as plant species richness (Banaszak, 1980; Steffan-Dewenter and Tscharntke, 2001) or the abundance of plants needed for food and reproduction (Thomas et al., 2001). Different pollinating insects are influenced by factors operating over different spatial scales. Small bee species are dependent on floral resources close to the nest (Gathmann and Tscharntke, 2002; Vulliamy et al., 2006), whereas larger bees may often fly further for floral resources (Westrich, 1996).

Plant assemblages on isolated or fragmented habitat sites may in turn have amplified ecological pressures on their ongoing survival. Where low population densities exist there can be a knock-on effect of decreased fitness owing to the decreased chance of genetic out crossing (Holt et al., 2004). With animal pollinated plants, a small population size of resource plants may be less attractive to pollinators, resulting in fewer visitations and so further decreasing the likelihood of pollen transfer (Cheptou and Avendano, 2006). The resulting disproportionate loss of viability than would have otherwise been expected from decreased density alone is the 'Allee effect' (Allee et al., 1949). Depending on the extent of the Allee effect, assemblages of plants and insect pollinators can be disrupted and may even become extinct (Cheptou and Avendano, 2006).

The importance of restoration

Published definitions of restoration include:
The process of returning a degraded system to a healthier more "normal" state (Falk et al., 2006).
and:
The process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (SERI, 2004).

The goal of restoration is to create a habitat which is self-sustaining without the need for human intervention. Within this thesis the term 'restoration' is used rather than habitat creation, reclamation or rehabilitation. The term 'restoration' for habitat creation is used within the landfill industry and has links to aftercare use and policy input considerations (Davis and Coppeard, 1987; Robinson and Handel, 1993; Tosh et al., 1994; Basri, 1998; Simmons, 1999; Rebele and Lehmann, 2002; Gregory and Vickers, 2003; Kim et al., 2004; Remon et al., 2005; Athy et al., 2006). Whilst landfill sites may not necessarily be returning to their previous land use, as implied by 'restoration', they are being restored to an after use befitting their locally evolving landscape; be it grazing or semi-natural vegetation (Watson and Hack, 2000).

The main objectives of restoration in post-industrial landuse include: terrain stabilization, future public safety, aesthetic appeal, and the return of the land to a useful purpose within its regional context (Davis and Coppeard, 1987; Gilbert and Anderson, 1998; Simmons, 1999). This restoration often has an ecological based approach. Restoring habitats is important to mitigate the effects of intensification of agriculture and urbanisation. The effects of increasing human population, urbanisation and intensification may lead to fewer natural habitats and pollinating insects, but with restoration of wild habitats this cycle of degradation may be mitigated (Figure 1.01). It is an issue which can be seen in two ways: pollinators are important to restoration since many species of plants rely on insect mediated pollen transfer in their reproduction; but restoration or conservation sites are important to pollinators facing threats to habitats.

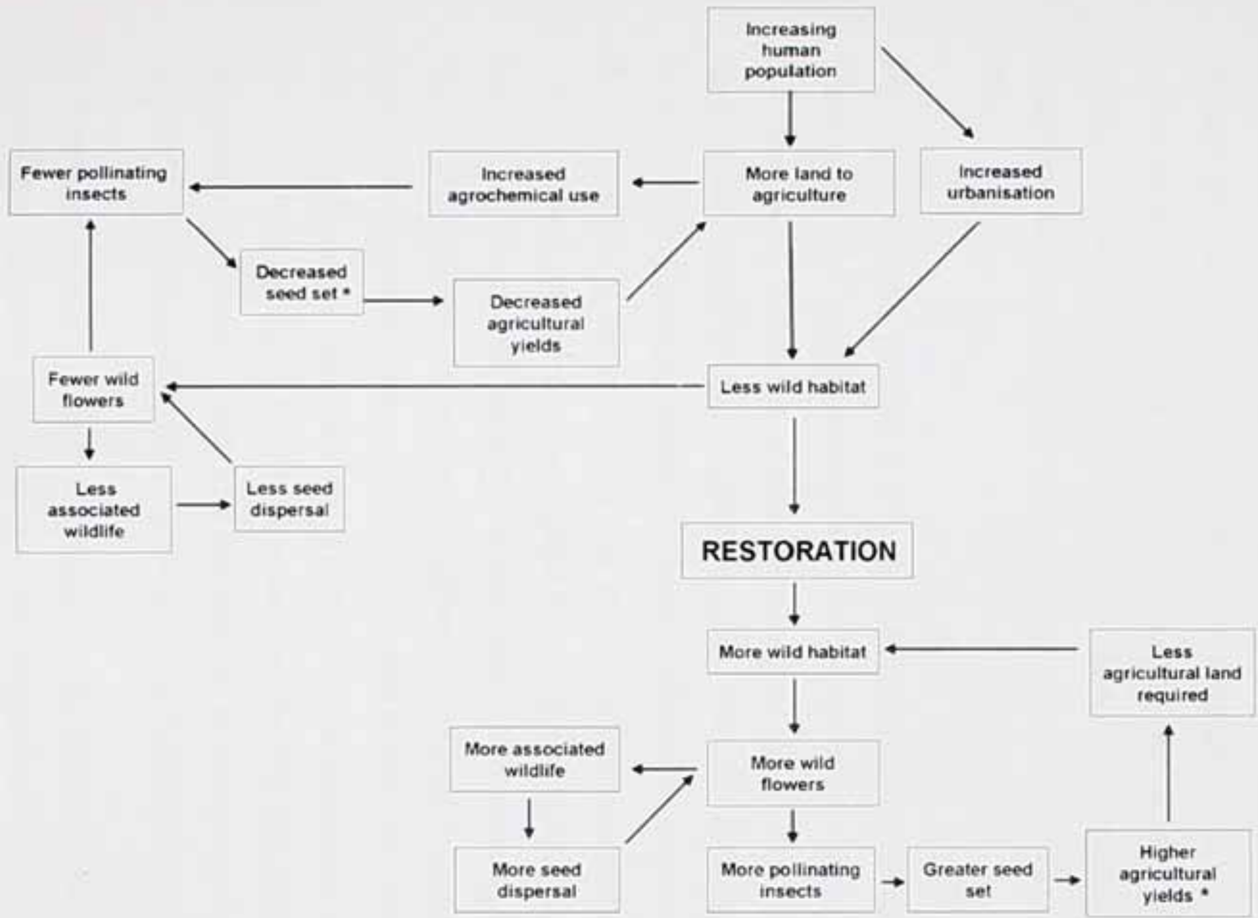


Figure 1.01 Pollinating insects in relation to decreasing habitat and mitigation through restoration. * Assumes crops may be pollen limited with regards 'Decreased seed set' and 'Higher agricultural yields' in figure. (Allen-Wardell et al., 1998; Goulson, 2003b; Kremen et al., 2004; Steffan-Dewenter et al., 2005; Greenleaf and Kremen, 2006; Morandin and Winston, 2006; Klein et al., 2007; Kasina et al., 2009).

Landfill restoration and the waste industry

There are approximately 2,200 working landfill sites in England and Wales, covering 28,000 ha. and they are closing at a rate of about 100 per year (Environment Agency, 2006). For the most recent data available, in England and Wales in 2005, 119 million tonnes of waste were disposed of, and 60% went to landfill, 22% for treatment, 6% for incineration, and 10% went into metal recycling facilities (Environment Agency, 2007c). A total of 72 million tonnes of waste was deposited at landfill sites. This compares to 75 million tonnes in 2004/5 a reduction of around 5%. Overall landfill deposits have fallen by 15% since 2000/1. For the East of England region, nearly 11.7 million tonnes of waste was disposed of in landfill sites in 2005 (Environment Agency, 2007b). In total in England and Wales there was 685 million cubic metres of remaining landfill capacity at existing permitted sites in England and Wales on December 31st 2005.

Prior to the 1990 Environmental Protection Act, restoration and aftercare standards for landfill sites were highly variable, with little in the way of engineering systems (Watson and Hack, 2000). The extent of restoration on older sites can be very poor, with thin capping and limited pollution control measures. Since the 1990 Environmental Protection Act, landfill operators have been required to be responsible for after-use on their sites. There is currently a five year statutory aftercare period (Watson and Hack, 2000). Landfill site operators in England have a regulatory mandate to monitor gas and leachate production from landfill sites and report at least annually to the Environment Agency (HMSO, 2002).This statutory aftercare period and the procedural ecological impact assessment, has required landfill operators to consider specific habitat creation and the maintenance required for long term conservation benefit. The UK Biodiversity Action Plan set up targets for the conservation of habitats and species over the following years (Anon., 1994). This has been one of the driving forces behind the waste industry's restoration policy for closed landfill sites (Watson and Hack, 2000).

Standard contemporary practice for site closure depends on its waste composition, i.e. whether inert, domestic or hazardous waste. The practice for the most common type, domestic waste, involves capping off, using a synthetic liner or compacted clay, then overlaying sub-soil followed by top soil. The site is then hydro-seeded, a method of sowing where a liquid emulsion of seed, fertiliser and binding agent is sprayed onto the site (Watson and Hack, 2000). Hydro-seeding has the advantage that it does not compact the soil with heavy machinery. Seed is typically bought in bulk from seed merchants. This method continues due to it being cost effective, low maintenance and increasing the likelihood of successful revegetation.

The characteristics of a modern restored landfill site (Figures 1.02 - 1.04), including its domed landform, compacted clay cap, and artificial topsoil, can present a difficult environment for habitat creation as they become dry in the summer and waterlogged in the winter (Watson and Hack, 2000; Rawlinson et al., 2004). The major outputs from a landfill site following decomposition processes are landfill gas and leachate. These outputs are at their peak during the first few years after land filling, but are contained within the engineering systems (Farquhar and Rovers, 1973; Watson and Hack, 2000). These gases can be vented off, but are usually collected and burned to produce electricity. In 2006, energy from waste contributed approximately one third of renewable energy in the UK (Stiles, 2008).

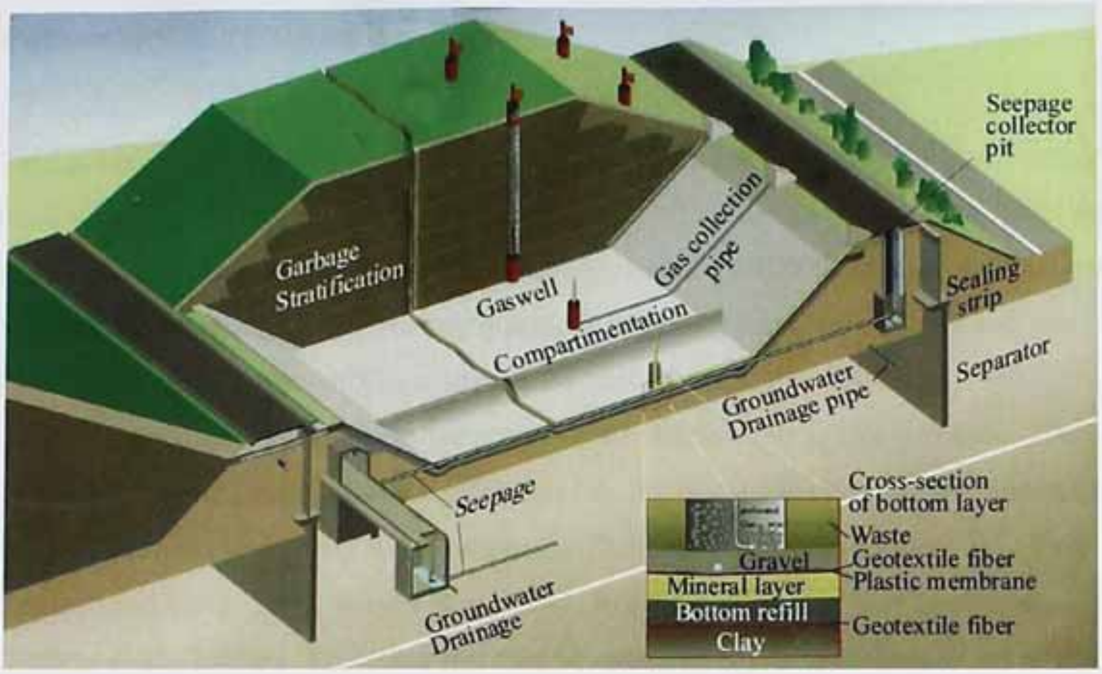


Figure 1.02 – A modern landfill site, showing engineering structures and capped profile (Makel Engineering, 2006).

Figure 1.03 – Example of a restored landfill site – Wootton.

Scale: image= 800m horizontal, 900m vertical 52°11'26"N 0°53'28"W

Pollinators in restored habitats.

Pollinators are important vectors in plant reproduction; however they do have complex habitat requirements which may affect their ability to colonise restored sites. They also have differing dispersal abilities and flight ranges which affect their dispersal between habitats. Those groups of pollinating insects with greater dispersal abilities and less specific habitat requirements have greater colonisation abilities (Tscharntke et al., 2002). For example, bumblebees are not generally habitat specialists (Goulson, 2003a), and their foraging ranges and hence dispersal ability are at least several hundred metres and up to several kilometres from their nests (Dramstad, 1996; Osborne et al., 1999; Goulson and Stout, 2001; Kreyer et al., 2004; Knight et al., 2005; Westphal et al., 2006; Osborne et al., 2008b). Therefore it would be expected that bumblebees disperse easily across the landscape and colonise or utilise new habitat sites readily. Those pollinating insect groups with poorer dispersal abilities and more specific habitat requirements have poorer colonisation abilities (Tscharntke et al., 2002). There is little information regarding the dispersal abilities of non-bee pollinator groups; however, butterflies often require specific host plants for their larvae and have been found to be species poor in fragmented habitats, suggesting that they have poor dispersal abilities (Steffan-Dewenter and Tscharntke, 2000; Krauss et al., 2003).

Pollinators can be important for plant ecology restoration as they enable fertilization and hence seed production. Seeds are important as they allow plants to escape and disperse spatially and in time, something that vegetative or clonal growth rarely does (Fenner and Thompson, 2005). Seeds allow annual plants to re-grow the following years. The effects of a healthy pollinator community increases seed and fruit production, providing valuable food resources for a variety of frugivore wildlife including birds, small mammals and other insects.

Studies concerning restoration ecology have indicated the importance of proximity to local natural habitat for restoration sites, being reservoirs of colonising and later successional species (Handel et al., 1994; Montalvo et al., 1997). Pollinators are important for the success of a restoration project and must colonise from nearby natural habitats or be deliberately introduced. Semi-natural areas provide habitats for populations of pollinators and resources to pollinators that live in the surrounding landscape (Steffan-Dewenter et al., 2002). Conserving fragments of semi-natural habitat within intense agricultural areas can benefit biodiversity generally and improve crop productivity (Tscharntke et al., 1998; Ghazoul, 2005; Morandin and Winston, 2006). Increased semi-natural habitats in a regional area can promote diversity and abundance for butterflies, bees and hoverflies (Steffan-Dewenter et al., 2002; Kleijn and van Langevelde, 2006). Greater distances from semi-natural habitats negatively affect the species richness and abundance of pollinators on crops (Steffan-Dewenter et al., 2005), and in parts of the world where there are decreased pollinator populations this has resulted in reduced plant reproduction (Kevan et al., 2002).

Conservation initiatives within the agricultural landscape are often aimed towards restoring and conserving linear semi-natural vegetation features such as hedgerows and field margin strips, often being subsidised through agri-environmental schemes (Banaszak, 1980; Pywell et al., 2005; Morandin and Winston, 2006). In deteriorated agricultural environments the provision of semi-natural areas can have a greater effect than semi-natural linear features (Kleijn and van Langevelde, 2006). Pollinator insect movement and the resulting gene flow may be diminished by distance and fragmentation, in particular for smaller populations of less rewarding flowering plants. Therefore rare species within the landscape which have few flowering resources will be affected most (Kwak et al., 1998; Courchamp et al., 2006).

The restoration of pollinator communities is a complex problem due to the multiple requirements of pollinators, requiring varying food and nesting materials and sites (Table 1.01). The relationship between habitat restoration and flower-visiting insects, as stated previously may be seen in two ways, relating to the potential of these sites to support flower-visiting insect conservation efforts, and for the potential of these mutualistic interactions to illustrate the underlying functioning of the habitats and how successful the restoration of habitats has been.

Studies have examined the positive role that restored sites can have for flower-visiting insect conservation. Roadside verges are one such resource of land not competing with agricultural land use, which provide floral resources. A study on prairie restoration on road verges in North America, found that those sites with greater abundance and richness of native flowers supported a greater bee abundance and species richness (Hopwood, 2008). Another valuable source of land is reclaimed opencast mines; again these are another source of land not competing with agricultural use. Butterfly communities and floral resources were assessed on 18 reclaimed mine sites in North America (Holl, 1995). The sites were found to provide 300 times the nectar abundance of the surrounding land use, but this decreased with time following reclamation. Butterflies' species richness and abundance was significantly correlated with floral resources, but they require a number of habitat variables which must be considered in their conservation. These studies highlight the potential value of restoring substantial areas of land benefiting pollinator conservation efforts.

In Britain research relating to the restoration of pollinator communities has been conducted in hay meadows by Forup and Memmott (2005b). They found that in comparing two newly restored hay meadows with two older sites, the restoration process had been successful since although there was some structural variation between the sites, the pollen transport webs showed the same level of connectance between the old and new sites. However, it should be noted that the level of conspecific pollen deposition on plant stigmas was not measured, and hence true levels of pollination were not assessed. Studies of pollinator interaction on restored heathlands have compared four pairs of restored and ancient sites (Forup et al., 2008). Restoration was determined as successful through similar species richness, although there was little species overlap between the sites. The interaction networks were less complex, with regards to connectance for the restored sites. They highlight how a key ecological process can be a valuable yardstick for measuring the success of restoration.

This research project is of importance as it improves our relatively limited knowledge relating to flower-visiting insects within the restoration process; regarding both the conservation benefits for insects and their use in determining successful restoration. No studies have examined the pollinator community on restored landfill sites in the UK.

The outcomes from this research may influence both the landfill operators and government policy, to create more flower-visitor friendly habitat sites, and so bolster the conservation effort for pollinating insects. The research will also further our knowledge in using the assessment of community structural elements to determine successful restoration.

General aims

This research project comes at an appropriate time as pollinating insects are suffering population declines. The restoration of habitats is of great importance due to our current land use policy and farming practice in the UK. If restored landfill sites can support a viable and abundant assemblage of pollinating insects then they have great potential for conserving these species. This study will further our knowledge related to plant and pollinating insect assemblages within restoration practice and its success in relation to interaction structure.

The broad goal of this research project is to examine the potential of restored landfill sites to support flower-visiting insect assemblages; this will be shown through the following chapters answering the research questions as shown below.

1. What are the floristic characteristics of restored landfill sites?

This focuses on the insect pollinated flowering plants, their abundance, richness and the difference in species composition between the restored and reference sites. The sites will be surveyed over the whole blooming season, allowing determination of the phenological development. Answering this question will specify the floral resources available for the flower-visiting insects.

2. How successful is landfill site restoration for flower-visiting insects and what effects do habitat quality variables have?

This question will focus on those flower-visiting insects found feeding on the restored landfill sites, their abundance, species richness, and the difference in species composition compared to reference sites. Answering this will indicate the potential conservation value of the restored landfill sites for flower-visiting insects.

3. **How is the assemblage of flower visitors and plants structured? Is it different on restored sites compared to reference sites?**

The structure of the plant-insect interactions illustrates the functioning of the restored habitats. This may show the success of restoration and the robustness of the newly created habitat.

4. **Can these results be used to understand the broader factors relating to the potential of restored landfill sites to support flower-visiting insects within the landscape? What recommendations can be made for landfill site operators with regards to restoration practice?**

This question will give an indication as to the management of the restored landfill sites within the landscape context, how restoration processes have enabled the pollination assemblage and future practices which restoration practitioners can use to encourage plants beneficial for flower-visiting insects.

Figure 1.04 Brixworth landfill site - 52°20'32"N 0°53'23"W

Scale: image= 1.2km horizontal, 1.5km vertical

Chapter

2

General Methods

General Methods

"Reason, observation, and experience; the holy trinity of science."
Robert Green Ingersoll (American Statesman. 1833-1899)

"That's the whole problem with science. You've got a bunch of empiricists trying to describe things of unimaginable wonder."
Bill Watterson (b.1958), "Calvin and Hobbes"

Introduction

This chapter includes the general methods used in this thesis and a critical appraisal of them. Specific collection and analysis methods are covered within the separate research question chapters. The data for this study were from original fieldwork collection.

Selection of Study sites

Restored Landfill Sites

The research has been conducted on sites in the East Midlands of the UK, in the counties of Northamptonshire, Bedfordshire, Warwickshire and Buckinghamshire. All of the sites are within 50 km of Northampton. Originally, a total of 42 landfill sites with licenses to dispose of waste were identified. These sites were then surveyed for suitability of use in this project. The initial survey described the sites in terms of accessibility, size, age and disposal type. The aim was to select sites of similar characteristics to limit the number of variables having to be considered, whilst remaining representative of restored landfill sites as a whole. Following this, nine sites where chosen (Figure 2.01 and Table 2.01). The restored landfill sites had a minimum distance of 6 km between them. The mean foraging distance for flower-visiting insects is below 4km, and as such the sites can be considered as independent samples (Osborne et al., 1999; Gathmann and Tscharntke, 2002; Greenleaf et al., 2007; Osborne et al., 2008b).

a)

b)

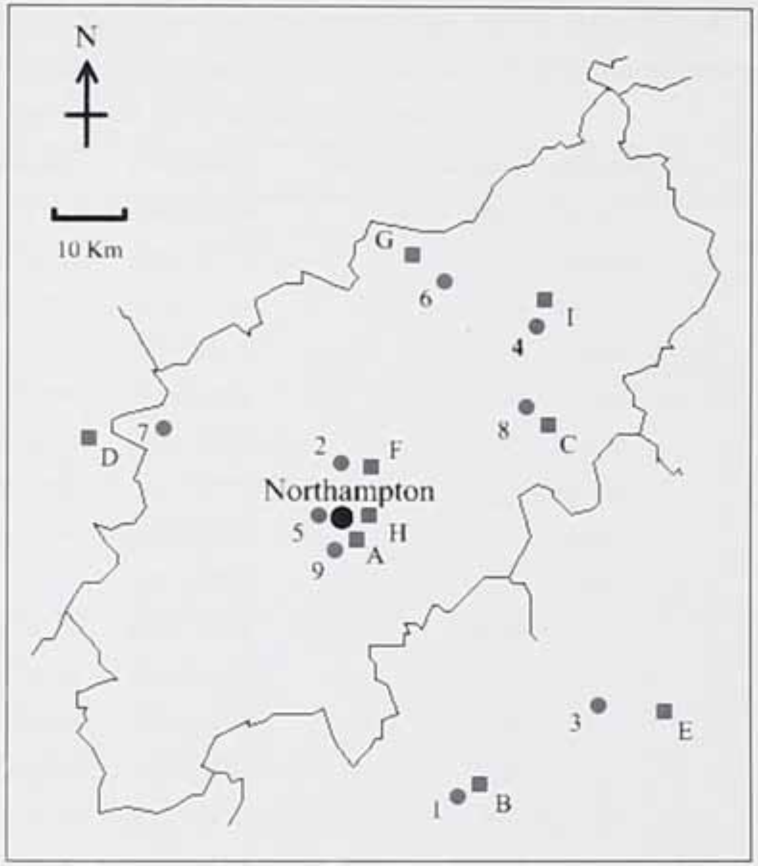


Figure 2.01 Maps of: a) Study region within the UK, Counties outlined, b) Selected study sites: ● (1-9) restored landfill sites, ■A-I reference sites.

1. Bletchley, 2. Brixworth, 3. Brogborough, 4. Cranford, 5. Harlestone, 6. Kettering, 7. Kilsby, 8. Sidegate Lane, 9. Wootton.

A. Barnes Meadow, B. Blue Lagoon, C. Ditchford, D. Draycote, E. Glebe Meadow, F. Pitsford, G. River Ise Meadows, H. Scrub Fields, I. Twywell.

Table 2.01 Details of restored landfill sites and reference sites selected. Status refers to landfills being Working / Closed or Reference site designation. Landfill Operators – WRG = Waste Recycling Group, Sita = Sita UK, BPL = Barton Plant Limited, Biffa = Biffa Waste services and Viridor = Viridor Waste Management. References sites – WT – The Wildlife Trust for Bedfordshire, Cambridgeshire, Northamptonshire and Peterborough. MKBC – Milton Keynes Borough Council, WWT – Warwickshire Wildlife Trust, BCC – Bedfordshire County Council, Age (Years after restoration) and Size (Hectares) refers to the restored areas or reference site connected area. [1] – Age (years) as of 2009 following restoration. Revegetation method: sown or naturally colonised.

Site	Type	Status	Operator	Latitude	Longitude	Age[1]	Size (ha.)	Revegetation	Pairing	Data
Bletchley	Landfill site	Working	WRG	51°58'57"N	0°45'26"W	5	34.00	Sown	1	2007
Brixworth		Closed	Sita	52°20'32"N	0°53'26"W	8	11.25	Natural	2	2007 / 2008
Brogborough		Working	WRG	52°02'44"N	0°35'31"W	5	26.11	Sown	3	2007
Cranford		Working	Sita	52°22'31"N	0°37'50"W	4	0.58	Natural	4	2007
Harlestone		Working	BPL	52°16'01"N	0°57'53"W	4	6.60	Natural	5	2007
Kettering		Closed	BPL	52°25'34"N	0°43'09"W	15	10.80	Natural	6	2007
Kilsby		Working	Biffa	52°19'00"N	1°10'07"W	12	7.19	Natural	7	2007
Sidegate Lane		Working	Sita	52°19'35"N	0°39'23"W	15	4.13	Sown	8	2007 / 2008
Wootton		Closed	Viridor	52°11'26"N	0°53'28"W	7	14.57	Sown	9	2007 / 2008
Barnes Meadow	Reference site	LNR	WT	52°13'44"N	0°52'23"W	18	4.18		9	2007 / 2008
Blue Lagoon		LNR	MKBC	51°59'20"N	0°44'19"W	40+	1.05		1	2007
Ditchford		LNR	WT	52°18'03"N	0°38'10"W	40+	12.19		8	2007 / 2008
Draycote		SSSI	WWT	52°20'00"N	1°20'20"W	30+	2.25		7	2007
Glebe Meadow		LNR	BCC	52°03'17"N	0°28'34"W	24	4.12		3	2007 / 2008
Pitsford		SSSI	WT	52°19'38"N	0°51'32"W	30+	0.79		2	2007
River Ise Meadows		SSSI	WT	52°26'11"N	0°44'17"W	20	21.36		6	2007
Scrub Fields		LNR	WT	52°16'06"N	0°52'48"W	12	3.00		5	2007
Twywell		LNR	WT	52°23'09"N	0°37'11"W	30+	8.08		4	2007

Nine landfill sites were surveyed in the first year (2007) of the study accompanied by paired reference sites. Nine sites were selected as this was considered an appropriate sample size, allowing surveying each season within the blooming period. The following year (2008), four landfill sites accompanied by paired reference sites were surveyed, although this was reduced to three due to logistical issues (Brixworth Landfill site / Pitsford SSSI, Sidegate Land Landfill site / Ditchford LNR, Wootton Landfill site / Barnes Meadow LNR). The four sites were randomly selected from the nine sites from the previous year, and were therefore representative of them. The reduced number of sites was used in the second year to build up a more comprehensive flower visitor catalogue.

Reference sites

Once the restored landfill sites had been selected, appropriate comparison reference sites were chosen. Reference sites were used to enable species baseline comparisons. The reference sites were selected as the closest site of recognised nature value, being designated as either Local Nature Reserves (LNR) or Sites of Special Scientific Interest (SSSI). (NB. In cases where sites have been designated for non-ecology reasons it is recognised that they would still be managed in an ecologically sensitive manner.) Site designations and locations were identified using the Natural England map tool (Natural England, 2006). Local nature reserves can be used as a comparison to measure the success of restoration, and that invertebrate communities may be accurate determinants of ecological function (Handel et al., 1994).

The rationale for selecting nearby designated conservation sites as reference sites was that such sites provide a potential 'end-point' or 'target' in terms of ecological restoration. The reference sites would be close enough to the landfill sites so they would experience similar local climates, have the same regional plant and insect species pool, and be found in similar landscape contexts (Figure 2.01). Reference sites, were more appropriate than control sites given that there were no environmental effects from the landfill sites themselves. Traditional methods of using control sites are limited; using a reference sites approach instead allows the restored sites to be compared to the means from all the reference sites. In laboratory experiments the use of 'controls' with all variables monitored or manipulated is possible (Reynoldson et al., 1997). In field experiments all variables cannot be controlled. The reference sites will therefore be exposed to the same species pools and landscape contexts as the restored landfill sites.

Site selection criteria

From previous research and knowledge of the character of landfill and reference sites available, the following selection criteria were determined. This design thus aims to control undue variation in plant and flower-visiting insect abundance and diversity. This is to ensure truer comparisons are made, and so representative landfill sites are chosen.

Restored landfill site selection criteria:

- **Restored area should be at least 50% of the whole site.**

This is to avoid undue influence of the restored habitat from ongoing landfill site operations. Also reduce health and safety issues.

- **Minimum size of 0.5 ha.**

To ensure sites are typical full scale landfill sites.

- **Have been designated as restored for at least 4 years.**

So that flowering plants have become established onsite and so flower-visiting insects may be present.

- **Be within 50km of Northampton.**

So all sites are within the same regional landscape and for logistical travel reasons.

- **No significant detrimental factors observed.**

To ensure sites being surveyed are not atypical for their floristic characteristics and flower-visitor assemblages.

Reference site selection criteria:

- **Minimum size of 0.5 ha.**

To ensure sites are of a similar size to the restored landfill sites.

- **Closest area of designated land matching the set criteria.**

To ensure both the restored landfill and reference sites have the same landscape context.

- **Site must not be undergoing any specific pollinating insect conservation measures.**

To allow for a fair realistic comparison.

- **Similar vegetation type to that of the restored sites i.e. grassland.**

To ensure no large scale differences in the plants surveyed or their associated flower-visitors.

Whilst the sizes of the paired restored landfill and reference sites will be different, the equal area of surveying, described below, aims to remove the island-bio-geographic effect on species richness. Surveying each site proportionally to their area was considered but rejected as it would make equal effort sampling difficult to achieve.

The research design for this study enables a direct assessment of the restored sites in comparison to their reference sites. This has been achieved through pairs of restored and reference sites having the same landscape context, removing the conflicting effects of the landscape context and habitat quality. This has been confirmed through research indicating no significant differences within the landscape context, between pairs of sites, or between types of site (Rahman, 2009). In addition, there were no substantial areas of comparable nature sites found within the measured radius of the landscape context (up to 1000m from the centre of each site). There was also no difference between the landscape context of different sites, Northamptonshire and the surrounding region being quite homogeneous in its rural land use, dominated by mixed farming.

Field work

Timing of fieldwork

The fieldwork season ran from March to October as this corresponds to the main flowering period in central England (Fitter et al., 1995). Flowering times in the British Isles usually start in early spring and continue until late autumn. The original plan was for all sites to be each surveyed three times during the fieldwork season (Table 2.02). This was to cover the early, mid and late flowering species. However during the 2007 field season, summer rain affected the rate of field work, and so whilst all the sites where surveyed twice, only two pairs were surveyed three times. For 2008, given the reduced subset the intention was to sample sites with greater intensity. The research design involved a balance between the number of sites studied (sample size) and the detail in which each site can be surveyed. In order to establish broad generalities about sites it is often advantageous to sample a larger number of sites in the first year of the study and then use these results, focusing on fewer sites to address specific questions, in the second year. This was achieved, although in early spring one of the reference sites was sampled only once; however the effect of this should be minimal as there were very few flowers this early and hence there would have been little flower visitor activity then, and a mid-spring survey was conducted later.

The sites were randomly sampled in pairs, so the same sites were not always sampled first or last. There were however, constraints placed upon this random selection, since it was desired that the restored landfill sites and their reference sites will be sampled on consecutive days to reduce temporal bias. In addition, due to restricted access, the restored landfill sites could not be surveyed at weekends and required two days notice prior to fieldwork. These constraints were built into the sampling programme and fieldwork was undertaken whenever the weather conditions permitted.

Table 2.02 Fieldwork summary, indicating number of days per season spent surveying on each site for 2007 and 2008.

Spring = before 1st of June, summer = 1st June – 31st August, autumn = after 31st August.

Type of site	Site	2007			2008		
		Spring	Summer	Autumn	Spring	Summer	Autumn
Landfill	Bletchley	1	1	1			
	Brixworth	1	1	1	2	1	1
	Brogborough	1		1			
	Cranford	1		1			
	Harlestone	1	1				
	Kettering	1	1				
	Kilsby	1		1			
	Sidegate Lane	1		1	2	1	1
	Wootton	1	1	1	2	1	1
Reference	Barnes Meadow	1	1	1	1	1	1
	Blue Lagoon	1	1				
	Ditchford	1	1		2	1	1
	Draycote	1		1			
	Glebe Meadow	1		1			
	Pitsford	1	1	1	2	1	1
	River Ise Meadows	1	1	1			
	Scrub Field	1	1	1			
	Twywell		1	1			

Floral surveys

A standardised belt transect was used in the surveying of insect pollinated plants in flower. A random systematic approach was taken: the approximate centre of the sites were located and marked, a direction was chosen using randomised bearing tables, and used if the size or shape of the sites permitted. If it did not then a new direction was used from the bearing table. A 100m x 2m transect was laid out with its centre at the marked point. All flowering plants were identified to species level and number of flowering units recorded. One floral unit was defined as; a head (e.g. *Trifolium repens* (Figure 2.02)), an umbel (e.g. *Daucus carota*) or a capitulum (e.g. *Centaurea scabiosa*). On this issue, Dafni et al. (2005) state that as a rule of thumb, inflorescences should be described as pollination units if the distance between individual flowers is less than the size of the individual flowers. The belt transect method is an efficient way to survey a large area (Dafni et al., 2005). This method is a standard approach to plant surveys and has been used in previous studies (Dicks et al., 2002; Forup and Memmott, 2005a).

Figure 2.02 Example of the floral unit of White clover (*Trifolium repens*).

Identification was made using a standard British field guide to flowering plants, using the nomenclature of Stace and Kent (Stace, 1991; Kent, 1992; Rose and O'Reilly, 2006). Any plants which could not be identified in the field were collected and labelled for identification and pressing (Dafni et al., 2005). The floral surveys each sampled 200m^2 and were repeated twice per fieldwork day (i.e. sampled 200m x 2m per site per day). From herein 'flowering plants' refers to insect pollinated plants which are in flower.

Critique of this method highlights that transects were not fully random in their selection. i.e. they were always from the centre of each of the sites; this may have achieved a systematic error within the data, but as this technique was conducted on all of the sites, any bias should be minimal and equal sampling effort was applied to all of the sites. Secondly, the plant species found solely at the edge may be missed, or under represented, however from personal observations across the whole site this is not believed to be the case.

Floral resources

The floral cover method used in this study to represent floral resources, combines floral abundance with inflorescence size. The positive relationship between nectar production and inflorescence size and frequency is well established (Harder and Cruzan, 1990; Holl, 1995; Pacini et al., 2003). Inflorescence frequency has been used as indicative of sugar abundance (Sharp et al., 1974; Kremen, 1992; Munguira and Thomas, 1992; Forup and Memmott, 2005b; Forup et al., 2008), and corolla size is less susceptible to plant vigour and hence site specificity than nectar secretion rates (Harder and Cruzan, 1990). Floral cover is sufficiently robust to allow comparisons across sites and has been used in previous pollination restoration studies (Meyer et al., 2009) and relating to pollinating insect resources (Steffan-Dewenter and Tscharntke, 2001; Potts et al., 2006; Vulliamy et al., 2006; Clough et al., 2007; Holzschuh et al., 2007; Ebeling et al., 2008; Meyer et al., 2009). Estimation of floral nectar production per plant species was not undertaken due to constraints of research resources available and sample size (nearly 100 plant species altogether). Nectar also has great variation between individuals of any given plant species, both within and between sites and also with temperature, humidity, solar radiation, age, position of the plant, success of fertilization etc. (Corbet, 1978; Corbet et al., 1979; Tepedino and Stanton, 1982; Pleasants and Chaplin, 1983; Lanza et al., 1995; Pacini et al., 2003).

Mean cross-sectional surface area of the floral unit was calculated for each flowering plant species and then multiplied by their frequencies, to obtain a measure of floral cover. Floral units were used, i.e., umbels and inflorescence rather than single flowers. The sizes of the floral units was determined through field sampling, and checked for consistency using field identification guides (Rose and O'Reilly, 2006), data from botanical journals e.g. *Alliaria petiolata* (Cruden et al., 1996), *Cirsium eriophorum* (Tofts, 1999), and Knuth's handbook on flower pollination (Knuth, 1906-1909) (for flower sizes – see Appendix 1). Only open flowers likely to produce nectar or pollen were recorded. Significant differences were determined for the seasonal mean floral cover as cm^2 of floral bloom per m^2 of transect and total floral cover per site as m^2 of floral bloom per site.

Flower visitor insect surveys

Flower-visiting insect surveys were undertaken three times between 9am and 4pm on days which were warm and sunny with little or no wind, as outlined in the Butterfly Monitoring Scheme (Pollard and Yates, 1993) and similar to those used in previous pollination studies (e.g. Banaszak, 1980; Kearns and Inouye, 1993; Dicks et al., 2002; Goverde et al., 2002; Dafni et al., 2005; Forup and Memmott, 2005b; Kleijn and van Langevelde, 2006; Potts et al., 2006; Nielsen and Bascompte, 2007).

In 2007, flower visitors were surveyed along the transect line as defined in the floral survey section (Figure 2.03). The transect line was left undisturbed for 20 minutes following the initial entomophilous plant survey to allow the flower visitors to return. The 100m transect was then surveyed at a rate of approximately three metres per minute, giving enough time for observation and capture. The following year, 2008, the sampling followed a systematic pattern following a spiral from the approximate centre of the site, at a standard pace of 10 metres / minute. This was similar to the survey method used by Nielsen and Bascompte (2007) and determined to be effective for ecological surveys (Kalikhman, 2007). The spiral method employed allows for proportional sampling from all plant species in flower. Given a 2m wide transect, in each transect an area of approximately 600m^2 was surveyed. This method allowed for a larger area to be surveyed owing to the relatively low pollinator density, and so greater data accumulation. Transect methods have been used many times as a way to sample flower visitors over a large area where they may be relatively low in abundance (Banaszak, 1980; Fussell and Corbet, 1992; Lagerlof et al., 1992; Steffan-Dewenter and Tscharntke, 2001; Dicks et al., 2002; Forup and Memmott, 2005b; Greenleaf and Kremen, 2006; Potts et al., 2006; Nielsen and Bascompte, 2007; Forup et al., 2008).

Surveys lasted 30 minutes and all flower visiting insects seen to be feeding legitimately (i.e. not nectar robbing) and large enough to touch anthers and stigmas were captured from along a 2 metre wide belt and within 2 metres in front of the surveyor. All flower visitors were considered as 'pollinators' irrespective of their 'quality' (i.e. their effectiveness at transferring conspecific pollen during each visit). Any time spent in transferring captured insects to jars was deducted from the total time to achieve a constant sampling effort. No distinction was made between different types of feeding behaviour i.e. pollen versus nectar. The first species of plant which the insect was seen visiting was the one recorded. The pollinator survey was conducted on the transects at three different times (morning, midday and afternoon) to capture those insects active at different times.

The captured insects were then transferred individually into labelled plastic jars. After a survey was completed, those insects that could be identified in the field were recorded and released. Those insects that could not be identified were represented by voucher specimens taken for later identification. The samples were then identified using standard keys and reference collections for the Hymenoptera and Syphidae.

The transect method to survey flower-visiting insects has been used often, but it does have some issues related to it. The common assumption is that all flower visitors captured are potential pollinators (Kevan and Baker, 1983). The likelihood of the insect being a possible pollinator is increased since only those seen to be on or near the flower's anthers or stigmas were captured. This does not completely rectify the problem however since the insect needs to be carrying conspecific and viable pollen.

The technique relies on the observer to first spot the insect and then to capture it. Inevitably some insects will be missed by the observer, and the action of capturing with a net may scare off other insects further ahead. Although it is impossible to determine exactly the percentage of flower visitors missed, with experience levels of ≥90% capture are achieved, with the slow rate of transect work.

An alternative method was to select random squares (e.g. 5m x 5m) then observe flower visitors and record for 10 minutes e.g. (Sjödin, 2007). However, this gives a slower rate of observation, which would not be as good on low density of flowers, and where it is likely to miss rarer plant species. This alternative method does rely on the ability of the observer to recognise pollinators down to species level which would take considerable training and is actually impossible in the field for many cryptic taxa.

Habitat quality characteristics

This study will focus on habitat quality in terms of the importance of floral resource diversity, and the importance of other physical environmental factors for pollinating insect habitat. The local environmental variables attributing to habitat quality assessed included; area, vegetation height and density, bare earth and sand soil, microstructures, south facing slopes, hedgerows and shrubs, flowering plant and plant richness, flower abundance and the soil physical and chemical properties. For further details see Chapter 4 methods section.

Landscape context

Community studies often focus on habitat fragments in isolation, surrounded by a 'sea' of non-habitat, and ignore the importance of the surrounding landscape matrix (Ricketts, 2001). Landscape context influences pollinator diversity and abundance, and different pollinators respond at different ranges (Steffan-Dewenter et al., 2002). However, the effects are mixed and not consistent for species or groups of flower-visiting insects. For example, no straightforward relationship has been found for landscape context and bumblebee richness or abundance (Steffan-Dewenter et al., 2002; Westphal et al., 2003; Kleijn and van Langevelde, 2006). For butterflies the dominant factor is habitat size and no significant relationship with habitat isolation has been found (Krauss et al., 2003). Numerous studies examine the relative effects of habitat and landscape context on flower-visiting insects (Thomas et al., 2001; Backman and Tiainen, 2002; Hatfield and LeBuhn, 2007; Sjödin et al., 2008), and show that landscape context does play a part in determining the species of flower-visiting insects and their abundance on habitat sites.

As mentioned previously, the research design for this study enables a more direct assessment of the restored sites in comparison to their reference sites. This has been achieved through pairs of restored and reference sites having the same landscape context, removing the conflicting effects of the landscape context and habitat quality. This has been confirmed through research indicating no significant difference within the landscape context, within pairs of sites, or between types of site (Rahman, 2009). The effect of habitat quality is now possible to examine alone.

Statistical methods

Statistical methods have been discussed within the separate chapters. Where statistics are not presented within the text they are presented within figure legends.

Summary

The methods described above are typical to those used within flower-visiting studies in the landscape. They proved effective at gathering data for this thesis and there is no recognised significant bias to the data. The sample sizes were appropriate to be large enough for statistical analysis and considering the time frame and resources of this PhD study.

Figure 2.03 The author undertaking fieldwork on Cranford landfill site, 2007.

Chapter

3

The floristic characteristics of restored landfill sites

The floristic characteristics of restored landfill sites

"**Where flowers bloom so does hope**."
Claudia 'Lady Bird' Johnson, Public Roads: Where Flowers Bloom (1912-2007)

Summary

This chapter examines whether the revegetation of restored landfill sites is comparable to reference sites with regards to the insect pollinated flowering plant community. This has been assessed through direct comparison between restored landfill sites and reference sites using standard belt transect surveys to record plants in flower. These surveys were repeated throughout the flowering season to record the phenological development. Sites were also examined for the effects of soil characteristics and revegetation method.

There were one quarter of plant species found uniquely on restored landfill and reference sites but one half were found on both. There was no significant difference for the frequency with which species of plant were present or the number of species within a plant family on the restored landfill sites and reference sites. No difference was found between the restored landfill sites and the reference sites for the annual richness of plants in flower or abundance. Variation was found across the seasons and the restored landfill sites had lower species richness and floral abundance in the spring but higher in the autumn than the reference sites. The restored landfill sites that were sown had the same richness of plants in flower and floral abundance as those restored sites which were naturally revegetated. Examining soil characteristics showed differences between the restored landfill sites and the reference sites but no correlation with richness of plants in flower. The overall result shows that the landfill sites are being restored to a state comparable to that of the reference sites with regards to the insect pollinated flowering plant community.

Introduction

The importance of restoring areas of land to viable ecological sites is high due to anthropogenic causes of landscape degradation including habitat loss and agricultural intensification (Allen-Wardell et al., 1998). Landfill sites may be a valuable reserve of restorable land but there are certain factors which may affect the success of the restoration process and the species found on landfill sites. Landfill sites are typically restored using soils imported onto the site, which may be used as they are or mixed with improvers to meet the regulatory requirements of nutrient richness and homogeneity (Simmons, 1999; Watson and Hack, 2000). The soils are laid down using heavy plant machinery and then seeded or allowed to naturally colonise. The revegetated sites are then managed in a way to minimise the fire risk, with a summer mowing to remove excess vegetation. The end use of the sites is typically aimed at meeting planning requirements and not for conservation benefit (Simmons, 1999; Watson and Hack, 2000). There have been a number of studies investigating the potential of woodland establishment on old landfill sites (Ettala, 1988; Moffat and Houston, 1991; Robinson and Handel, 1993; Holley and Phillips, 1996; Rawlinson et al., 2004; Hutchings et al., 2006), but no research has been done relating to the insect pollinated flowering plants found on modern restored landfill sites in the UK.

Habitat restoration methods

The choice of which habitat type to create when restoring a post-industrial site should take into account a number of factors: the proposed end use, cost, ecological targets, land policy and environmental factors (Bell et al., 1997; Dobson et al., 1997; Basri, 1998; Hobbs and Harris, 2001; Falk et al., 2006). Possible end uses of restored sites may include public amenity space, wildlife habitat creation or agricultural use (Simmons, 1999).

Environmental factors which determine restoration vegetation choice include climate condition, slope and soil conditions and type (Chan et al., 1997). It is difficult or impossible in the long-term to grow vegetation that does not suit the prevailing soil conditions (Goodman, 1974). The most common problem with regards to soils and attaining a species-rich sward is fertility levels that are too high (Gilbert and Anderson, 1998).

Grassland restoration: seed mixes vs. natural colonisation

Grassland habitat creation is focused on here as this is most relevant vegetation type to this study, being the most prevalent landfill restoration vegetation type. There are three approaches to newly establishing flower-rich grassland: sowing a seed mix onto a newly tilled site, encouraging the diversification of an ecologically dull grass sward, or encouraging natural colonisation. The very limited grassland types now created on post-industrial sites in no way reflects the diversity of grasslands found historically in the UK (Gilbert and Anderson, 1998).

Off-the-shelf wild flower grassland mixes are available, with long catalogue lists of native species, and suites of species available to emulate NVC communities (Gilbert and Anderson, 1998). However, the cost of sowing a wildflower mix compares poorly with a basic grazing mixture and costs inevitably underlie decisions made by landfill operators (Table 3.01). Sowing wildflower grassland can be ten times cheaper than woodland creation but nearly 20 times more expensive than other grass types (Table 3.01). The cheaper options of seed mix, such as those used in landfill restoration, lack in floral resource species (Table 3.01).

Table 3.01 Different seed mixes commercially available - showing cost and species mix.

	Wild flower mix – Calcareous soil (A wild flower seed mix to be sown onto grassland)			Traditional long term grass ley Such as used on restored landfill sites			Department of Transport (DOT) Verge Mix	
Application	15 kg / ha.			35 kg / ha.			250 kg / ha.	
Cost per ha.	£2060 (MAS, 2008b)			£110 (MAS, 2008a)			£840 (Nicky's-Nursery, 2008)	
Species mix	Salad burnet	(*Sanguisorba minor*)	15.00%	Perennial ryegrass	Lasso	21.00%	Hard fescue	30.00%
	Self heal	(*Prunella vulgaris*)	15.00%	Cocksfoot	Sparta	18.00%	Perennial ryegrass	25.00%
	Ribwort plantain	(*Plantago laneceo*lata)	12.50%	Hybrid ryegrass	Solid	14.00%	Creeping red fescue	20.00%
	Ox-eye-daisy	(*Leucanthemum vulgaris*)	10.00%	Perennial ryegrass	Bree	14.00%	Smooth stalk meadow grass	10.00%
	Wild carrot	(*Daucus carota*)	10.00%	Timothy	Promes.	11.00%	Brown top bent	10.00%
	Black medick	(*Medicago lupilina*)	5.00%	Late Fl. red clover	Rajah	11.00%	White clover	5.00%
	Common vetch	(*Vicia sativa*)	5.00%	Mixed herbs		3.50%		
	Lady's bedstraw	(*Galium verum*)	5.00%	Alsike clover	Ermo	3.50%		
	Meadow buttercup	(*Ranunculus acris*)	5.00%	White clover	Alberta	3.50%		
	Musk mallow	(*Malva moschata*)	5.00%	Wild white clover	Nanouk	0.50%		
	Birdsfoot trefoil	(*Lotus corniculatus*)	2.50%					
	Corn poppy	(*Papaver rhoeas*)	2.50%					
	Cowslip	(*Primula veris*)	2.50%					
	Dropwort	(*Filipendula vulgaris*)	2.50%					

Natural colonisation

The natural colonisation of former industrial sites such as landfills relies on the recruitment of plant species from nearby sources. Vegetation colonisation may be through seed dispersal or tissue propagules. The question of whether there are enough sources of plant seeds or propagules within the local environment and of what species is important; prior to agricultural intensification, there would have been a greater source from local species rich grassland; however such sources have now declined. Local sources of vegetation could include field boundaries, field set-aside strips, areas of natural or semi-natural vegetation, and gardens within an urban/semi-urban environment. Sources of plant species may also come from the seed bank within the soil, which may have been sourced on-site or brought in. If the soils were local they may contain locally adapted species but if foreign, they probably will not. This leads to the increased likelihood of early colonisers now being predominantly agricultural pests or feral crop species and those of disturbed land, such as docks and thistles, rather than the desired grassland species. There is also the possibility of non-native invasive species taking over a site.

The most promising sites for natural colonisation are those with nutrient poor soils (Gilbert and Anderson, 1998). Landfill sites would potentially be suitable for this, as nutrient poor soils are easier and cheaper to obtain for restoration than nutrient rich ones. Often compost and soil 'improvers' are added to landfill top soils to meet environmental targets (Simmons, 1999; Watson and Hack, 2000). There is a cost benefit of using cheaper nutrient poor soils, and when this is combined with the cheapest revegetation option, then significant savings and so incentive is available. One possible hindrance could be that whilst re-colonisation is awaited then soil run off could occur, however with most opportunist species this would be minimal (Gilbert and Anderson, 1998). Another option sometimes used in trying to establish flower meadows is using non-aggressive annual grasses as a 'nursery' crop (Streever et al., 2000). The advantage of nutrient poor soils is that they provide a variety of niches for species to live in, without being outcompeted by more vigorously growing grass species. It has been found that patterns of plant richness in relation to nutrient levels generally follow a hump-backed-curve; species richness being low at low nutrient levels, increasing to a peak at intermediate levels, and declines more gradually with further increasing nutrient levels (Pausas and Austin, 2001).

Planting trees on restored landfill sites has been proposed (Ettala, 1988; Moffat and Houston, 1991; Crook, 1992; Robinson and Handel, 1993; Holley and Phillips, 1996; Hutchings et al., 2006). Tree planting on completed landfill sites is a viable alternative to grassland. However, fears remain of tree roots penetrating the impermeable cap and releasing landfill gases and leachate. This would also allow the ingress of water as well as oxygen beneath the surface potentially producing an explosive mix. Establishment problems were highlighted regarding the water-logging and drought conditions commonly occurring on restored sites (Moffat and Houston, 1991; Rawlinson et al., 2004; Hutchings et al., 2006). Woodland establishment on old-style landfill sites in North West England has seen to be effective, but environmental constraints were seen from soil depth and soil compaction (Rawlinson et al., 2004). The landfill operators however have shown little interest in the research and continued with grassland restoration.

On high quality soils, natural revegetation may not be as effective, particularly when there is an inadequate supply of native species locally. High fertility soils will colonise quickly from on-site seed bank and pioneer species, competitive growth will occur with weaker annual species being quickly eliminated, and a few species will dominate the site (Bayfield, 1995). Lower plant diversity is therefore likely. This may be desirable where a grazing end use is planned, with grasses dominating, but will be of lesser ecological value.

Post-creation remediation methods

Post-creation remediation is entered into to enrich an ecologically dull site, such as those on fertile soils. The remediation treatments required to convert fertile soil to a nutrient poor one may include nutrient stripping by crops or topsoil removal (Gilbert and Anderson, 1998). Topsoil removal could be potentially viable since it can be used elsewhere on-site; however, such treatments may be prohibitively expensive or labour intensive. Furthermore, topsoil on landfill sites may be an integral part of the engineered capping and structures such as gas and leachate wells would have already been integrated. On artificial laid soils, such as those on landfill sites, the topsoil would need to removed down to nearly the subsoil layer, this would be a substantial soil removal; for example a depth of 50cm being removed, would total $5000m^3$ / ha.

Flowering plants and floral resource provision

Flowers are diverse in their resource provision and reward pollinators through many different ways, typically through nectar and pollen rewards. Nectar is a very complex solution containing sucrose, fructose and glucose (Percival, 1961), some amino acids which insects cannot synthesise (Gardener and Gillman, 2002), lipids, vitamins and minerals. Other, perhaps rarer and more specialised resource provision by flowers, include attraction through scents, as in some orchids manufacturing bee sex pheromones, oil rewards, and some flowers mimic for visual stimulation (Buchmann and Nabhan, 1996).

The floral resources that entomophilous plants provide are crucial for supporting pollinators. Without floral resource provision within a habitat, flower visitors would be absent and grasses would dominate. Therefore, for the successful restoration of species rich plant and flower visitor assemblages on habitats the extent of floral resources is important.

Whilst it is true that most terrestrial restorations begin with seeding or transplanting, ultimately a restoration's long term ecological success relies on plant-animal interactions. To sustain most plant populations, reproduction is essential to replace individuals after their demise. Seed production is critical for regeneration of most plant populations and is essential if natural ecosystem dynamics are to occur on restored habitats. Most plant species depend on animal pollination for their reproduction and may disappear if their pollinators are not present (Powell and Powell, 1987; Steffan-Dewenter and Tscharntke, 1999). Insect-mediated pollination in turn contributes to the production of fruits and seeds that support diverse food webs (Bawa, 1990). On restored habitats, greater abundance and richness of plants in flower has been linked to greater abundance and richness of pollinating insects (Mortimer et al., 1998; Backman and Tiainen, 2002; Pywell et al., 2005; Hegland and Boeke, 2006; Franzén and Nilsson, 2008; Hopwood, 2008). Consequently both pollinating insects and flowering plants are interdependent, they rely on each other for either reproduction or resource provision and support numerous other organisms within habitats.

Aims

The aim of this chapter is to consider whether restored landfill sites have been revegetated in a similar way to reference sites for their insect pollinated flowering plant community. Flowering plant species richness and floral abundance will also be examined with regards to the effect of soil characteristics. The following questions will be addressed:

How does the richness of plants in flower and floral abundance (see Chapter 2) compare between restored landfill sites and reference nature sites? How does the richness of plants in flower and floral abundance compare between landfill sites which have been sown and those which have been naturally revegetated?

The expectation is that reference sites will have greater richness of plants in flower and floral abundance than the restored landfill sites. The restored landfill sites are newly created on artificial soils which may inhibit species richness. In addition, the soils are often compacted with heavy machinery. The sites may be sown with species poor seed mixes predominantly of ryegrass (*Lolium*) and clover (*Trifolium*). In contrast, the reference sites have been managed for the best possible ecological benefit by wildlife managers and have natural ecosystem dynamics occurring.

What is the effect of soil characteristics on richness of plants in flower on restored landfill sites?

It is expected that restored landfill sites and reference sites will have differences between them in soil characteristics including; bulk density, moisture, organic matter and stone content. It has previously been shown that richness of insect pollinated plants may decrease with increasing soil quality (Pausas and Austin, 2001). Therefore it is expected there will be a correlation between the richness of plants in flower and the soil characteristics measured.

How do restored landfill sites compare with the reference sites for the provision of resources for flower-visiting insects?

Flowering plant resources (principally nectar and pollen) are important to the pollinating insects found on the landfill sites as they provide the food for adults and larvae. Restored landfill sites will be considered for both their mean floral cover and also in relation to their impact upon the landscape by their total resource provision i.e. floral cover density x size of the restored sites. Floral cover has been used rather than total nectar and pollen resource availability as it is less weather and site specific, further rationale for this is discussed later in the Methods section.

Methods

Study region and study sites

The study was conducted in the East Midlands of the UK, in the counties of Northamptonshire, Bedfordshire, Warwickshire and Buckinghamshire. All of the sites are within 50 km of Northampton (see Chapter 2 - Figure 2.01). Nine landfill sites were surveyed in the first year (2007) of the study accompanied by paired reference sites. The following year (2008), four landfill sites accompanied by paired reference sites were initially surveyed, although this was reduced to three in the second field season due to logistical issues. For further details on-site selection criteria, see Chapter 2 – Methods.

Fieldwork timing

Fieldwork surveys were conducted from March to October, 2007 and 2008, as this corresponds to the main flowering period in central England and hence the flower visitor activity. Local weather conditions made uniform distribution of sampling days impossible. For distribution of survey days see Chapter 2 - Table 2.02.

Floral surveys

Standardised plant surveys were used (Dicks et al., 2002; Forup and Memmott, 2005; Potts et al., 2006). A random systematic approach was taken: the approximate centre of the sites were located and marked, a direction was chosen using randomised bearing tables, and used if the size or shape of the sites permitted. If it did not then a new random direction was used. A 100m x 2m transect was laid out with its centre at the marked point. All flowering plants were identified to species level and number of flowering units recorded. One floral unit was defined as a head (e.g. *Trifolium pratense*), an umbel (e.g. *Daucus carota*) or a capitulum (e.g. *Centaurea scabiosa*). Identification was made using a standard British flower field guide (Rose and O'Reilly, 2006). Two floral transects were surveyed per fieldwork day, with the data being combined and treated as a single dataset. At the intersection of the two transects, floral units were only counted on one transect to avoid double counting. From here in 'flowering plants' refers to insect pollinated plants which were in flower during these surveys.

Soil Analysis

Soil characteristics were recorded, namely: texture, bulk density, moisture, pH, organic and stone content (Table 3.02). All soil samples were collected in March 2009; pairs of sites were collected on the same day to enable true comparisons to be made. Five samples were randomly taken on each site from a depth of 10cm. These characters were assessed as they indicate soil quality, and so influence the plant community. Standard soil testing protocols were used (Anderson and Ingram, 1993; Rowell, 1994).

Soil texture was determined using a standard soil texture chart (Rowell, 1994) on fresh samples. Bulk density was determined using the weight of a fixed volume of sample, giving g cm^{-3}. This measure can give a crude measure of soil compaction, although it does not take type of soil into account. Moisture content was determined through drying of the sample for 24hrs in an oven at 105°C; the difference in weight was then used to determine percentage moisture (Rowell, 1994). The dried soil samples were then placed into a furnace at 500°C for 4 hours and re-weighed to give percentage organic content (Rowell, 1994). The stone content was determined through washing a known weight of soil through a 2mm sieve and then drying the resultant material.

Table 3.02 Soil characteristics and analysis methods.

Soil Test	**Rationale**	**Methods** (Rowell, 1994)
Texture	Soil type.	Observation and grain chart.
Bulk Density	Indication of compaction and hence soil quality.	Density via weighing known volume of soil.
Moisture Content	Consistency measure between sites.	Difference in weight following 24 hours at 105°C
Organic Content	Indicative of structure and age and so soil quality.	Difference in weight following 4 hours at 500°C
Stone Content	Structure and homogeneity and so quality of soil.	Washing a known weight of soil through a 2mm sieve. Dry weight of stones.
pH	Indicating chemistry of the soil and may affect plant community.	Soil sample agitated with de-ionised water and then tested using a calibrated probe.

Data Analysis

The data were tested for normal distributions using one-sample Kolmogorov-Smirnov tests. Levene's test was used to determine whether variances were significantly homogenous, and, if heterogeneous, the significance levels were adjusted accordingly. For testing differences between the types of sites within the pairing, paired samples t-tests and Wilcoxon signed rank tests were used to compare parametric and non-parametric data, respectively. For tests of differences in three independent samples, one-way ANOVAs or Kruskal-Wallis tests were used for parametric and non-transformable non-parametric data, respectively; post-hoc tests using paired samples t-tests were then used. For testing of difference in presence and frequency of species between types of sites paired t-tests and Wilcoxon signed rank tests were used to compare parametric and non-parametric data, respectively. Significance of correlations was determined using Pearson's correlations for parametric data and Spearman's rank correlations for non-parametric data. SPSS version 11.5 statistical software was used (SPSS, 2003). Abbreviations have been used: restored landfill and reference comparison sites may be referred to as 'landfill' and 'reference' respectively. Significant results: $p \leq 0.05$.

How does richness of plants in flower and floral abundance compare between restored landfill sites and comparison reference sites? How does richness of plants in flower and floral abundance compare between restored landfill sites which have been sown and those which have naturally revegetated?

To allow comparison between site types and show phenological development through the year the data were collated and then averaged for each of the seasons, giving flowering plant characteristics per site type in spring, summer and autumn. The first day of summer was taken to be the start of June and the first day of autumn the start of September. The restored landfill site data were compared for those sites which were sown and those naturally revegetated. Shannon diversity was analysed and the results are presented in Appendix 2. These were not included in the main thesis as the findings mirrored the species richness results. Restored landfill sites which were seeded and those naturally revegetated were identified from information supplied by the landfill operators.

Insect pollinated flowering plant species composition and floral abundance between types of sites are represented using non-metric multidimensional scaling (NMDS). Intervals in the data were measured using Euclidean distance. Euclidean distance was used rather than Bray-Curtis as it gives a greater distortion and hence visual spread to the data, which is an advantage when the sites being compared are expected to be very similar (Kessell and Whittaker, 1976). Unlike other ordination techniques NMDS does not make assumptions as to or about the distribution of the variables (Maarel, 2005). NMDS instead uses rank distances for ordination, and so this gives a visual representation with those sites having similar composition closer together (Legendre and Legendre, 1998; Maarel, 2005; Ollerton et al., 2009). For further information regarding use of NMDS see Maarel (2005).

What effect do soil characteristics have on species richness of plants in flower on restored landfill sites?

The soil characteristics means were compared between the restored landfill sites and the reference sites. Relationships between richness of plants in flower and soil characteristics were analysed. The richness of plants in flower was the cumulative total for the sites over the each of the years separately. To avoid pseudo-replication the flowering plant data were correlated against the means of the soil samples taken for each of the sites.

How do restored landfill sites compare with the reference sites for the provision of resources for flower-visiting insects?

Mean cross-sectional surface area of the floral unit was estimated for each flowering plant species and then multiplied by their frequencies, to obtain an estimate of floral cover. Floral units were used, i.e. umbels and inflorescence rather than single flowers. The sizes of the floral units was determined through field sampling, and ensured for consistency using field identification guides (Rose and O'Reilly, 2006), data from botanical journals e.g. *Alliaria petiolata* (Cruden et al., 1996), *Cirsium eriophorum* (Tofts, 1999) and Knuth's handbook on flower pollination (Knuth, 1906-1909) (for flower sizes – see Appendix 1). Only open flowers likely to produce nectar or pollen were recorded. Comparisons were made for the seasonal mean floral cover as cm^2 per m^2 and total floral cover per site m^2 between the restored landfill and reference sites.

Results

Over the duration of this study, an area of 25,000m^2 was surveyed for floral characteristics and approximately 138,000 floral units were counted from 98 plant species. On the restored landfill sites there were 63 species of flowering plant found in total and 19 exclusively found on the restored sites; on the reference sites there were 74 and 30 respectively. Sixty five species were common to both restored landfill and reference sites. There was no effect of site sizes on the flowering plant richness (2007 Pearson's correlation plant richness and size; restored landfill sites: $r = 0.21$, $p=0.58$, reference sites: $r = -0.49$, $p=0.21$).

The three most florally abundant plant species for each site have been identified (Table 3.03). The most common abundant species for the restored landfill sites, is *Trifolium dubium*; and for reference sites *Ranunculus acris* and *Galium verum*. *T. dubium* (Lesser Trefoil) is the smallest clover and very commonly found on grasslands in the British Isles (Rose and O'Reilly, 2006). *R. acris* (Meadow Buttercup) and *G. verum* (Ladies Bedstraw) are also both commonly found in grasslands throughout the British Isles (Rose and O'Reilly, 2006).

Table 3.03 The most florally abundant plant species on restored landfill and reference sites 2007.

Type	**Site**	**Most florally abundant plant species**		
		1st	**2nd**	**3rd**
Restored	Bletchley	*Lotus glaber*	*Trifolium dubium*	*Cardamine flexuosa*
	Brixworth	*Trifolium dubium*	*Galium aparine*	*Senecio erucifolius*
	Brogborough	*Picris echioides*	*Picris hieracioides*	*Trifolium dubium*
	Cranford	*Trifolium dubium*	*Lotus glaber*	*Picris echioides*
	Harlestone	*Trifolium repens*	*Cirsium vulgare*	*Picris hieracioides*
	Kettering	*Picris echioides*	*Cardamine flexuosa*	*Picris hieracioides*
	Kilsby	*Geranium dissectum*	*Taraxacum officinale*	*
	Sidegate Lane	*Trifolium dubium*	*Geranium dissectum*	*Ranunculus repens*
	Wootton	*Trifolium repens*	*Picris echioides*	*Senecio jacobaea*
Reference	Barnes Meadows	*Anthriscus sylvestris*	*Trifolium pratense*	*Trifolium repens*
	Blue Lagoon	*Vicia cracca*	*Melilotus officinalis*	*Lathyrus pratensis*
	Ditchford	*Ranunculus acris*	*Anthriscus sylvestris*	*Trifolium repens*
	Draycote	*Lotus corniculatus*	*Orchis morio*	*Ranunculus bulbosus*
	Glebe Meadows	*Trifolium dubium*	*Ranunculus bulbosus*	*Ranunculus acris*
	Pitsford	*Ranunculus acris*	*Stellaria graminea*	*Lotus corniculatus*
	River Ise Meadows	*Galium verum*	*Stellaria graminea*	*Trifolium repens*
	Scrub Fields	*Centaurea nigra*	*Trifolium campestre*	*Trifolium pratense*
	Twywell	*Galium verum*	*Trifolium dubium*	*Lotus corniculatus*

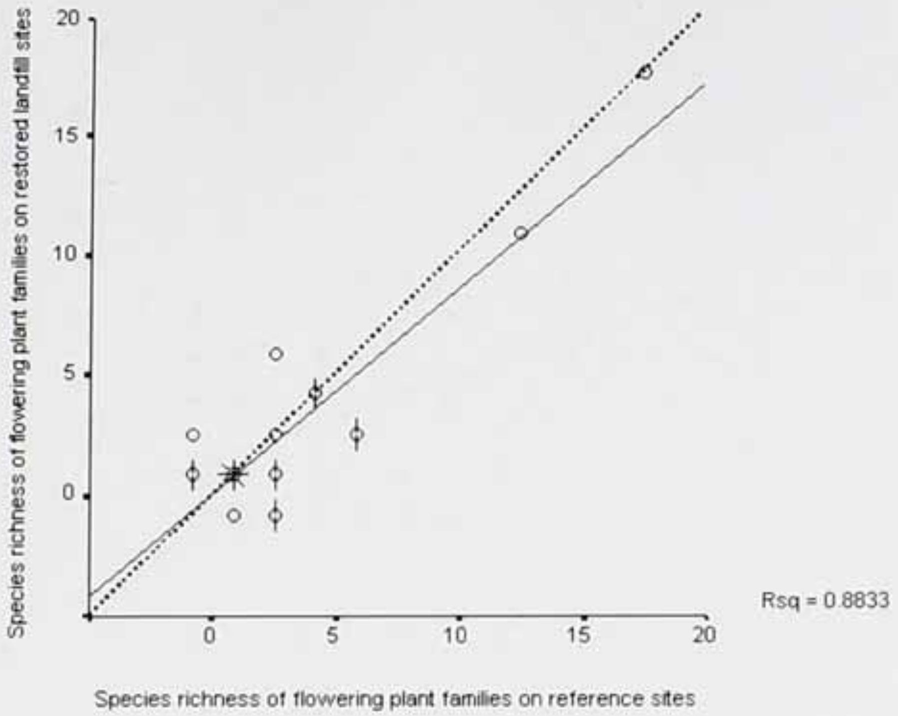
* only two species of plants in flower were found on Kilsby site.

There was significant similarity between the restored landfill and reference sites' species richness of the families of the flowering plants in 2007 and 2008 (Table 3.04 & Figure 3.01). Asteraceae was the most common family of plant and was found on both kinds of sites in both years. Asteraceae, also known as Compositae or commonly the Daisy family, is the world's largest family of flowering plants, and from the most common plant species found in this study contains: *Senecio* spp., *Cirsium* spp., *Centaurea* spp., and *Picris* spp. (Table 3.03) (Rose and O'Reilly, 2006). The species richness of families is also representative of their regional and national abundance (Figure 3.01).

The flowering plant species were analysed for frequency and presence by site. There was no significant difference between the species distribution on the two types of site in 2007 and 2008 (Figure 3.02) (Wilcoxon signed rank test (Two-tailed); 2007: $z = -0.58$, $p = 0.56$, 2008: $z = -1.021$, $p = 0.307$). The NMDS ordination plot of the species richness and abundance, shows that the majority of sites overlap, clumping together (Figure 3.03). This method is sensitive to showing outliers and the distance between points shows the relative similarity (McCune and Grace, 2002; Ollerton et al., 2009). The NMDS S-stress value below 0.1 (S=0.028) is small and therefore shows that the data similarity is truly represented by the 2 dimensions (McCune and Grace, 2002). This again illustrates the similarity in floral characteristic on restored landfill and reference sites. The few outlying sites result when a single species of high floral abundance is found almost uniquely on that site. For example, Glebe Meadow reference site (E) was abundant with *Ranunculus bulbosus* which was only found on one other site, Kettering restored landfill site (6) was abundant with *Cardamine flexuosa* found on one other restored landfill, and Pitsford reference site (F) was abundant with *Ranunculus acris*. Of real interest is how floristically similar 12 of the 18 restored landfill and reference sites are.

Table 3.04 Species richness of families of flowering plants on restored landfill and reference sites for 2007 and 2008. RL = Restored landfill sites, RF = Reference sites. Plant families are ordered by species abundance on restored landfill sites in 2007.

	2007		2008	
Family	**RL**	**RF**	**RL**	**RF**
Asteraceae	10	6	17	18
Fabaceae	8	6	10	13
Ranunculaceae	3	4	4	4
Boraginaceae	2	0	1	1
Rosaceae	1	2	3	6
Caryophyllaceae	1	2	1	3
Scrophulariaceae	1	1	6	3
Geraniaceae	1	1	1	1
Orchidaceae	1	0	1	2
Brassicaceae	0	1	4	4
Lamiaceae	0	1	3	5
Apiaceae	0	1	2	3
Rubiaceae	0	1	1	1
Onagraceae	0	0	2	0
Clusiaceae	0	0	1	1
Dipsacaceae	0	0	1	1
Gentianaceae	0	0	1	0
Liliaceae	0	0	1	0
Linaceae	0	0	1	1
Polygonaceae	0	0	1	1
Primulaceae	0	0	1	1
Convolvulaceae	0	0	0	2
Plantaginaceae	0	0	0	1
Violaceae	0	0	0	2
Total species richness	28	26	63	74



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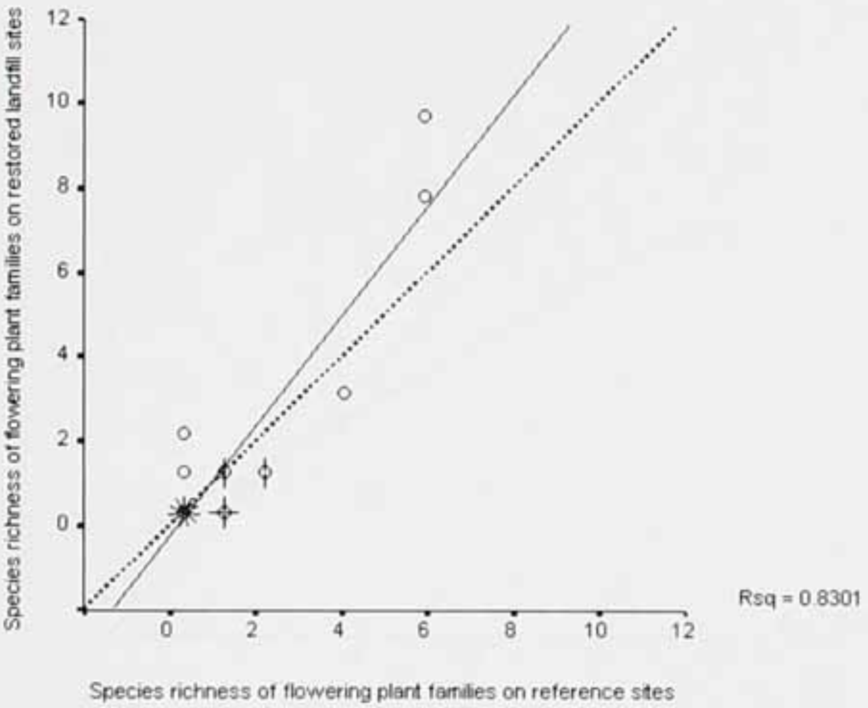
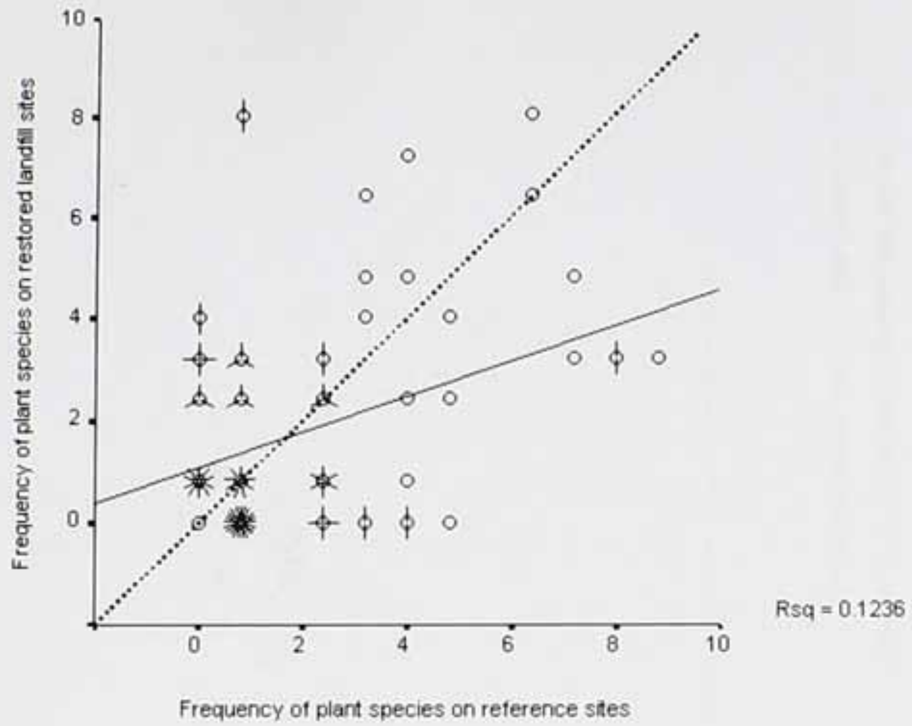


b)

Figure 3.01 Species richness of flowering plant families found on restored landfill and reference sites for: a) 2007, and b) 2008, Dashed line = 1:1, solid line = line of best fit. Spearman's rank correlation (Two-tailed); 2007: r = 0.623, p = 0.001, 2008 : r = 0.650, p = 0.001. Each data point is a family (See Table 3.04). 'Sunflowers' indicate multi-points, each petal represents an additional data point.



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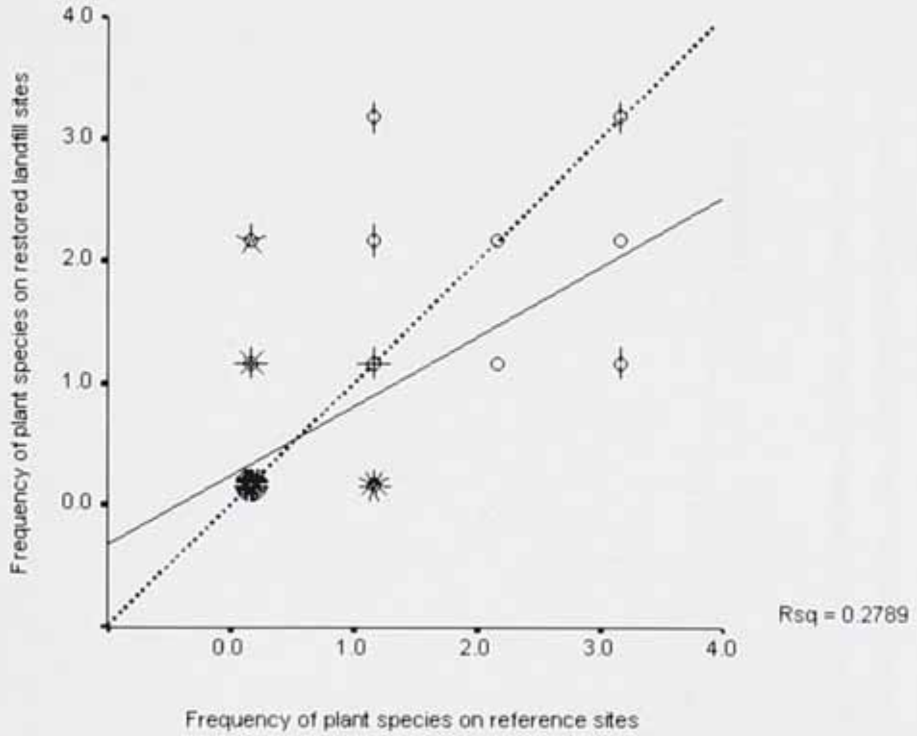


b)

Figure 3.02 Frequency of flowering plant species present on restored landfill and reference sites a) 2007 b) 2008 . Dashed line = 1:1, solid line = line of best fit. Each data point is one plant species. 'Sunflowers' indicate multi-points, each petal represents an additional data point.

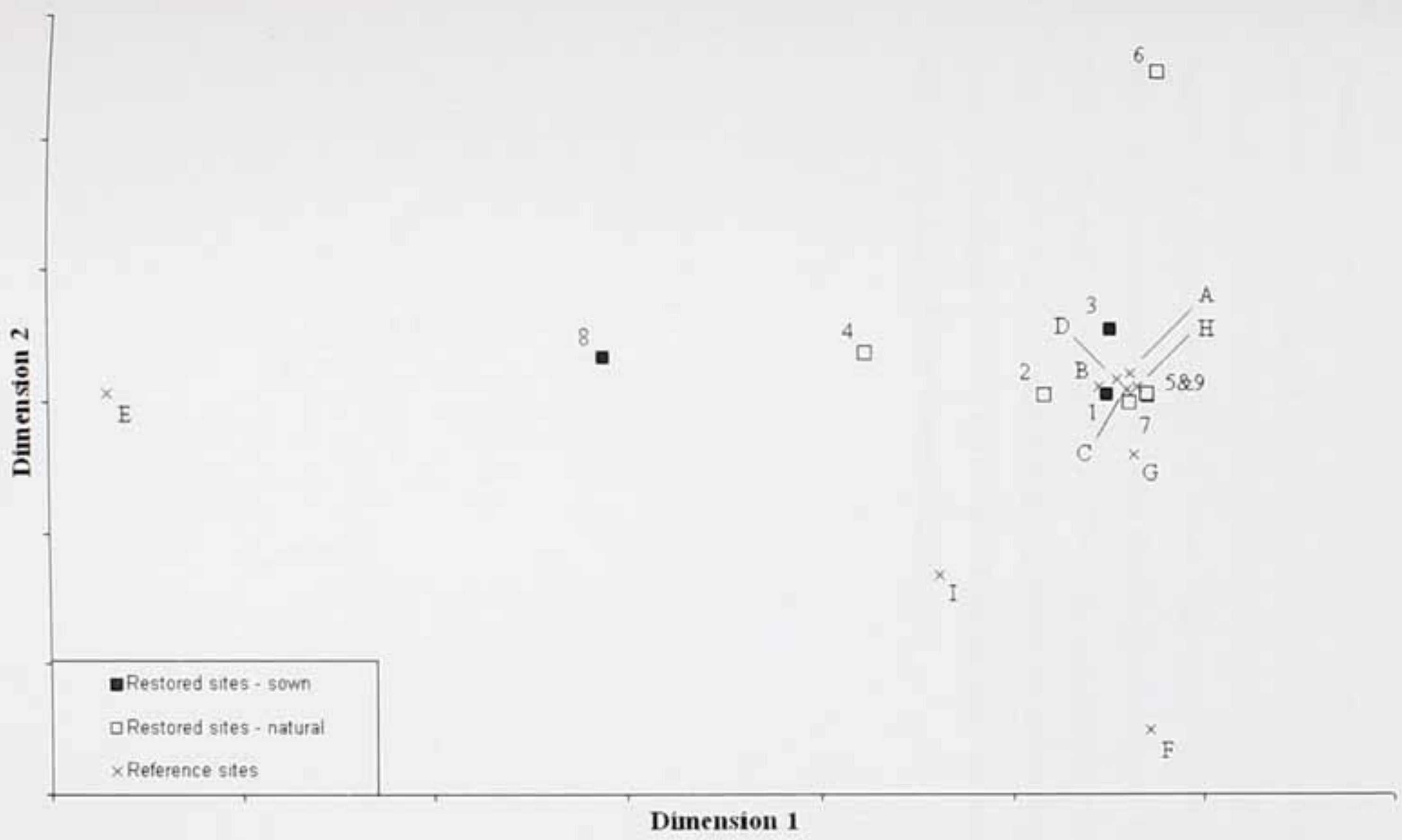
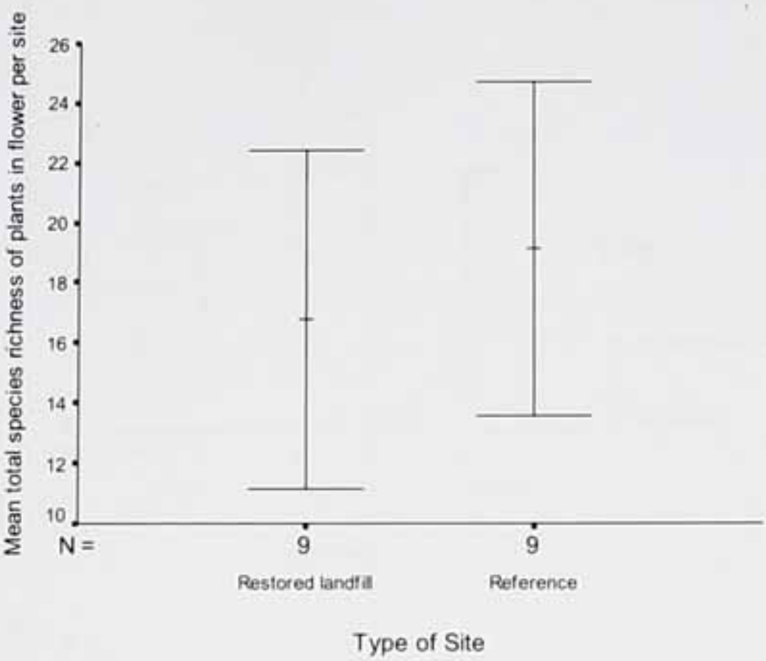


Figure 3.03 NMDS Ordination plot of plant species and abundance for sown and naturally revegetated restored landfill site (1-9) and reference sites (A-I) for 2007. Two-dimensional S-stress = 0.028. Restored landfill sites: 1. Bletchley, 2. Brixworth, 3. Brogborough, 4. Cranford, 5. Harlestone (natural), 6. Kettering, 7. Kilsby, 8. Sidegate Lane, 9. Wootton (sown). Reference sites: A. Barnes Meadow, B. Blue Lagoon, C. Ditchford, D. Draycote, E. Glebe Meadow, F. Pitsford, G. River Ise Meadows, H. Scrub Field, I. Twywell.

How does richness of plants in flower and floral abundance compare between restored landfill sites and reference sites? How do landfill sites which have been sown compare to those which have naturally revegetated?

Richness of plants in flower

There was no significant effect of site type on the annual 2007 and 2008 mean total species richness of plants in flower per site (Figures 3.04). There was a significant seasonal effect on the 2007 mean species richness of plants in flower for restored landfill and reference sites (Figure 3.05a). The restored landfill sites were significantly lower than the reference sites in mean flower species richness for spring but not for summer and autumn (Figure 3.05a). There was a significant seasonal effect on the reference sites' mean richness of plants in flower for 2008, but not for the restored landfill sites (Figure 3.05b). The autumn mean species richness of plants in flower was significantly higher for the restored landfill sites.



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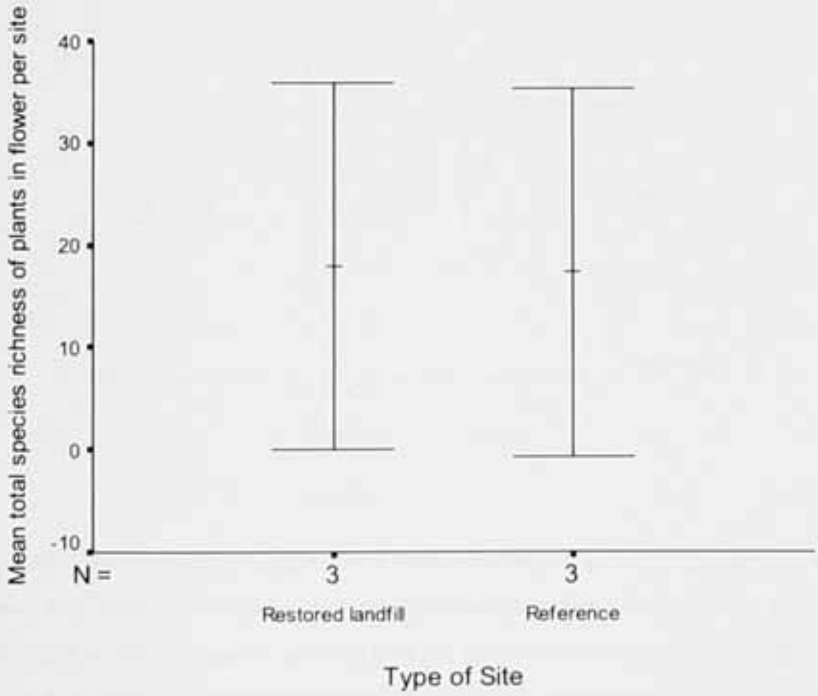
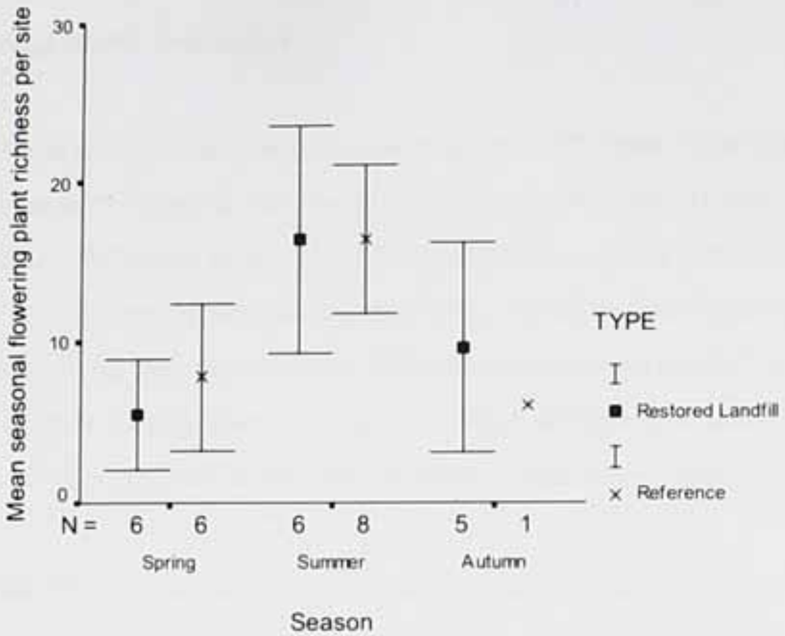


b)

Figure 3.04 Mean total species richness of plants in flower for restored landfill sites and reference sites (±95% Confidence Limits). N=Sample sizes. a) 2007 Paired samples t-test (two-tailed) $t=-0.94$, $df=8$, $p=0.37$, b) 2008 Paired samples t-test (two-tailed) $t=0.20$, $df=2$, $p=0.86$



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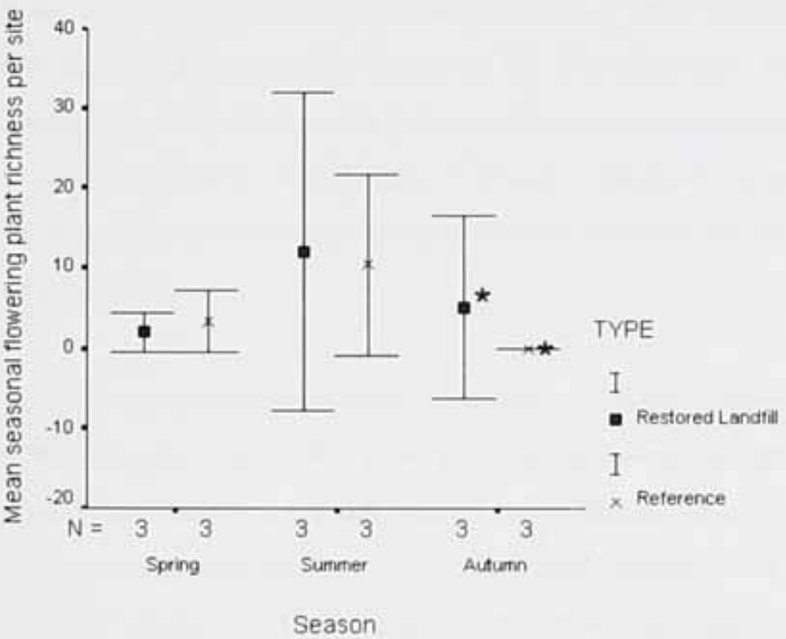


b)

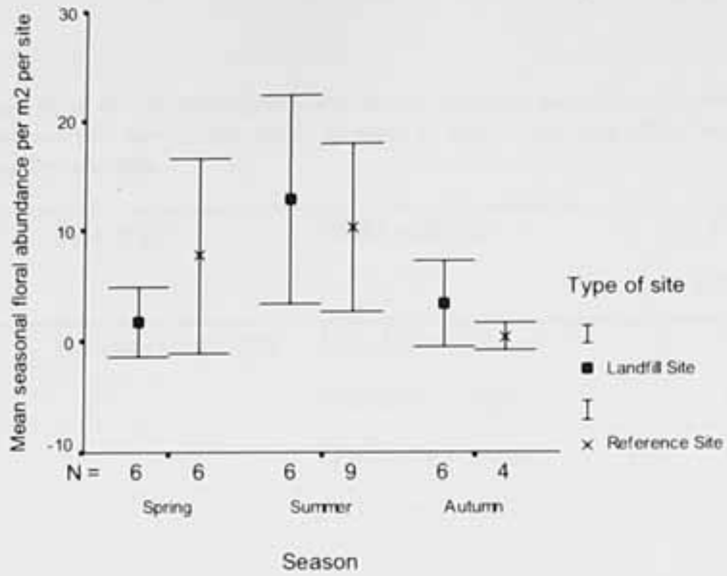
Figure 3.05 Mean seasonal species richness of plants in flower for restored landfill sites and reference sites (±95% Confidence Limits). N=sample sizes. * = medians. a) 2007 One-way ANOVA; Landfill sites across seasons $F_{2,16}$=6.40 p=0.01, Reference sites across seasons $F_{2,14}$=5.68 p=0.02. Paired Samples t-test (two-tailed) Spring Landfill vs. Reference t= -0.34 df=3 p=0.75, Summer Landfill vs. Reference t=0.45 df=5 p=0.68, Autumn Landfill vs. Reference na b) 2008 One-way ANOVA; Landfill sites across seasons: $F_{2,8}$=2.76, p=0.14, Reference sites across seasons: $F_{2,8}$=11.04, p=0.01. Paired Samples t-test (two-tailed); Spring Landfill vs. Reference: t= -1.51, df=2, p=0.27, Summer Landfill vs. Reference t=0.43, df=2, p=0.70. Wilcoxon test (two-tailed): Autumn Landfill vs. Reference Z=-1.34 p=0.18.

Seasonal floral abundance

There was a significant seasonal effect on the 2007 mean floral abundance per m^2 per site for restored landfill sites but not for the reference sites (Figure 3.06a). There was no significant difference between the floral abundance per m^2 for the restored landfill and reference sites in any season (Figure 3.06a). There was no significant seasonal effect on the mean floral abundance per m^2 for either the restored landfill or reference sites in 2008 (Figure 3.06b). There was no significant difference between the floral abundance per m^2 for the restored landfill and reference sites in any season (Figure 3.06b)

For individual plant species the five most abundant ones have been identified for the restored landfill and reference sites in 2007 and 2008 (Table 3.05). The most abundant species for 2007 is *Trifolium dubium* for both types of sites. Of note is that it is at a higher mean floral abundance per survey on the reference sites. For 2007, there are no other species of plant shared in the five most abundant between restored landfill and reference sites. For 2008, the only shared abundant species between both sites is *Lotus corniculatus*, which is more than twice as florally abundant on the reference than on the restored landfill sites.

There is a notable species and abundance difference in the most florally abundant species between 2007 and 2008 (Table 3.05). As there was no change in sampling time or method, this therefore potentially relates to the identity of sub-set of sites for 2008. The three sites randomly chosen for more intensive assessment in the second year had different species most florally abundant than the nine sites from the first year. The floral abundance was also lower in 2008, which was likely related to Sidegate Lane restored landfill (surveyed both years) having a low floral abundance.



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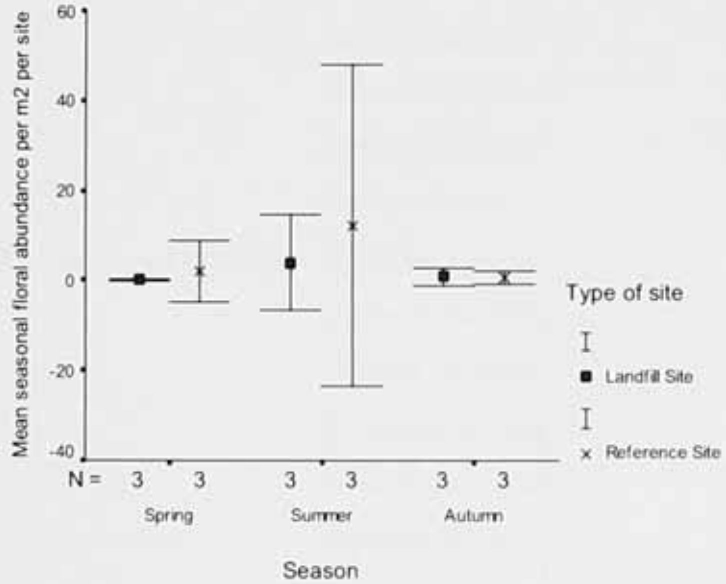


b)

Figure 3.06 Mean seasonal floral abundance per m^2 per site for landfill sites and reference sites (±95% Confidence Limits). N= sample sizes. a) 2007 One-way ANOVA: Landfill sites across seasons $F_{2,17}$=6.12, p=0.01, Reference sites across seasons $F_{2,18}$=1.91, p=0.18. Paired samples t-test (two-tailed) Spring Landfill vs. Reference t= -92 df=4 p=0.41, Summer Landfill vs. Reference t=0.76 df=5 p=0.48, Autumn Landfill vs. Reference t= -0.35 df=1 p=0.79 b) 2008 One-way ANOVA; Landfill sites across seasons: $F_{2,8}$=2.00, p=0.22, Reference sites across seasons: $F_{2,8}$=1.72, p=0.26. Paired samples t-test (two-tailed); Spring Landfill vs. Reference: t= -1.14, df=2, p=0.37, Summer Landfill vs. Reference t=-1.27, df=2, p=0.33. Autumn Landfill vs. Reference t= 2.00 df=2 p=0.18

Table 3.05 Most abundant plant species on restored landfill and reference sites. Mean floral abundance per survey: the mean number of floral units per $200m^2$ survey of a particular plant species for all sites.

Year	Site type	Plant species	Floral abundance	Frequency on sites
2007	**Restored landfill**	*Trifolium dubium*	337.6	6 of 9
		Picris echioides	153.8	8 of 9
		Trifolium repens	133.1	6 of 9
		Picris hieracioides	67.6	8 of 9
		Cardamine flexuosa	66.2	2 of 9
	Reference	*Trifolium dubium*	483.3	7 of 9
		Ranunculus bulbosus	162.0	5 of 9
		Lotus corniculatus	148.2	7 of 9
		Ranunculus acris	138.5	8 of 9
		Galium verum	135.5	2 of 9
2008	**Restored landfill**	*Trifolium repens*	120.5	3 of 3
		Lotus corniculatus	98.6	1 of 3
		Cirsium arvense	60.4	2 of 3
		Trifolium dubium	52.8	3 of 3
		Ranunculus repens	32.7	3 of 3
	Reference	*Ranunculus acris*	376.5	3 of 3
		Lotus corniculatus	253.5	2 of 3
		Stellaria graminea	135.6	1 of 3
		Galium verum	121.4	1 of 3
		Rhinanthus minor	114.2	1 of 3

How does the richness of plants in flower and floral abundance compare between those restored landfill sites which have been sown and those naturally revegetated?

For the mean total species richness of insect pollinated flowering plants there was no difference between the naturally revegetated and sown sites for 2007 (Figure 3.07). For the mean seasonal floral abundance per m^2 per site on sown and naturally revegetated landfill sites for 2007, there was no significant effect of the revegetation method for any season (Figure 3.08).

The sown and naturally revegetated restored landfill sites were assessed for flowering plant species and abundance using NMDS ordination (Figure 3.03). Six of the nine sites are closely clustered. There are three outliers, two naturally revegetated and one sown. Of the natural outlier sites, Cranford had abundant *Lotus glaber* (Narrow-leaved birds-foot trefoil) found little elsewhere, and Kettering, as stated previously, was abundant with *Cardamine flexuosa* found on one other restored landfill. The sown outlier site, Sidegate lane, was abundant with *Geranium dissectum* (Cut-leaved-cranes bill).

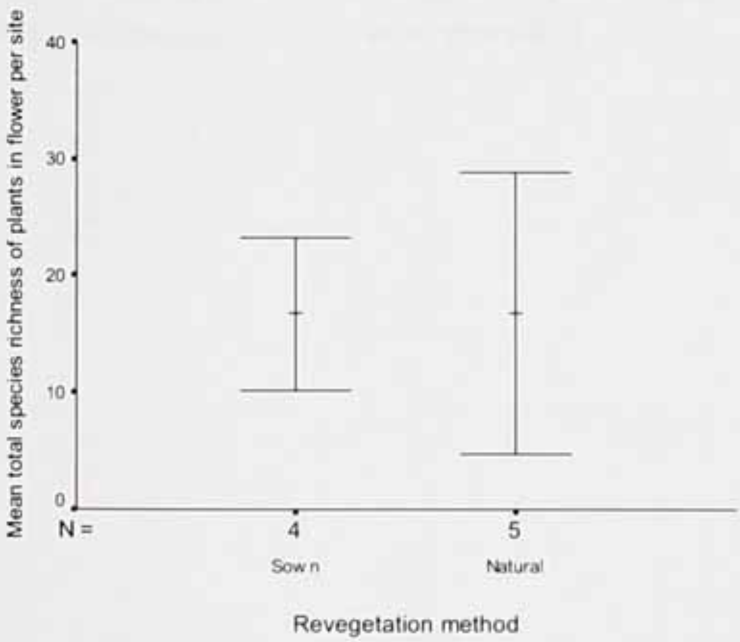


Figure 3.07 Mean total species richness of plants in flower on sown and naturally revegetated landfill sites in 2007 (±95% Confidence Limits). N= sample sizes. One-way ANOVA: $F_{1,8}$<0.001 p= 0.99.

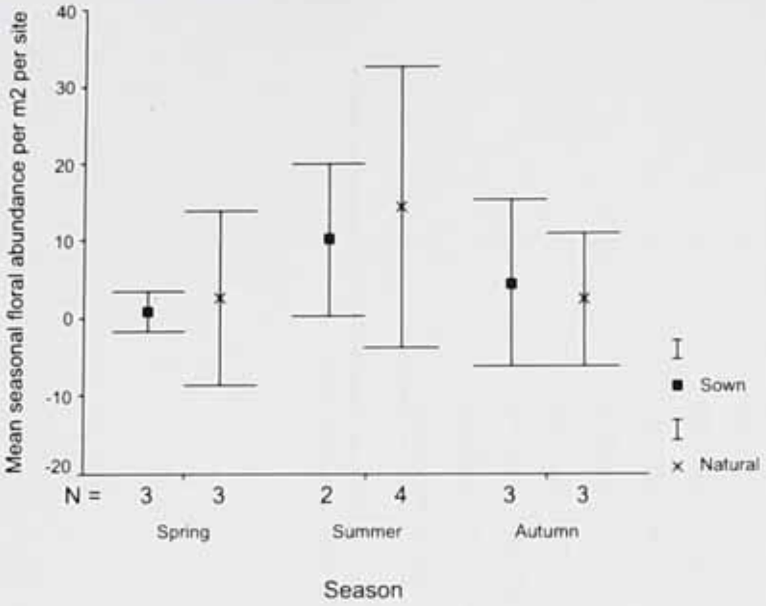


Figure 3.08 Mean seasonal floral abundance per m² per site on sown and naturally revegetated landfill sites 2007 (±95% Confidence Limits). N=Sample sizes. Independent samples t-test (two-tailed); Spring Sown vs. Natural: t= -0.65, df=2.22, p=0.58, Summer Sown vs. Natural t=-0.48, df=4, p=0.65. Autumn Sown vs. Natural t= 067 df=4 p=0.54

The results for this part of the study are summarised in Table 3.06.

Table 3.06 Summary of richness of plants in flower and floral abundance on restored landfill sites and reference sites.

Temporal scale	Variable	Year	Significance	Result
Annual	Mean richness of plants in flower	2007	n.s.	No significant difference
Seasonal	Mean richness of plants in flower	2007	n.s.	No significant difference
Annual	Mean richness of plants in flower	2008	n.s.	No significant difference
Seasonal	Mean richness of plants in flower	2008	n.s.	No significant difference
Seasonal	Mean floral abundance	2007	n.s.	No significant difference
Seasonal	Mean floral abundance	2008	n.s.	No significant difference
Annual *	Mean richness of plants in flower	2007	n.s.	No significant difference
Annual *	Mean floral abundance	2007	n.s.	No significant difference

n.s. – not significant, * – Sown and naturally revegetated restored landfill comparisons

What effect do soil characteristics have on the species richness of plants in flower found on restored landfill sites?

Soil samples were collected from each of the restored landfill and reference sites in March 2009 and analysed for bulk density and organic, moisture and stone content.

Comparing restored landfill and reference soil characteristics.

For the sites soil texture and pH there was no difference. For the pH: restored landfill sites: mean=7.25, min.=5.89, max=8.64, reference sites: mean=7.19, min.=5.71, max.=8.83; Mann-Whitney test of antilog pH: U=980.50, p=0.80. The majority of soils were clay silt loams. There were significant differences between the restored landfill and the reference sites in the other soil characteristics. The restored landfill sites had greater mean bulk density and stone content (Figures 3.09 & 3.12) and lower mean moisture and organic content than the reference sites (Figures 3.10 & 3.11).

The soil analysis shows that the restored landfill sites have poorer quality of soils being more compact, drier, containing less organic material and more stone.

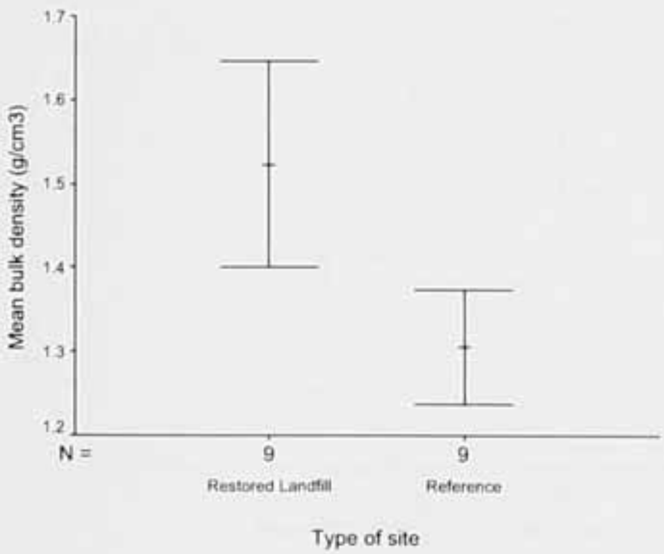


Figure 3.09 Mean bulk density of soils per sample (g cm^{-3}) for restored landfill and reference sites (±95% Confidence Limits). N=Sample sizes. Paired samples t-test (two-tailed); Landfill vs. Reference: t=5.65, df=8, p<0.001.

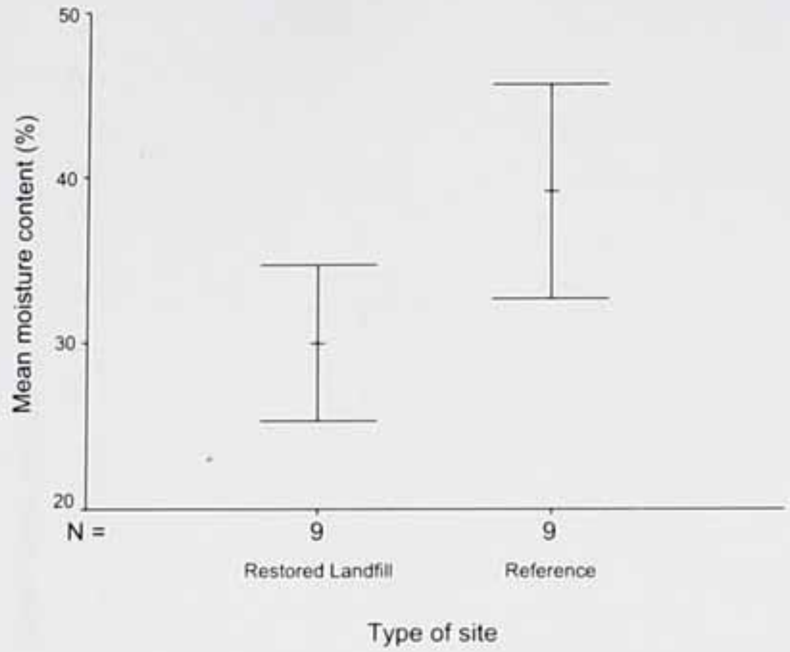


Figure 3.10 Mean moisture content per sample (%) for restored landfill and reference sites (±95% Confidence Limits). N=Sample sizes. Paired samples t-test (two-tailed); Landfill vs. Reference: t=-3.99, df=8, p=0.004.

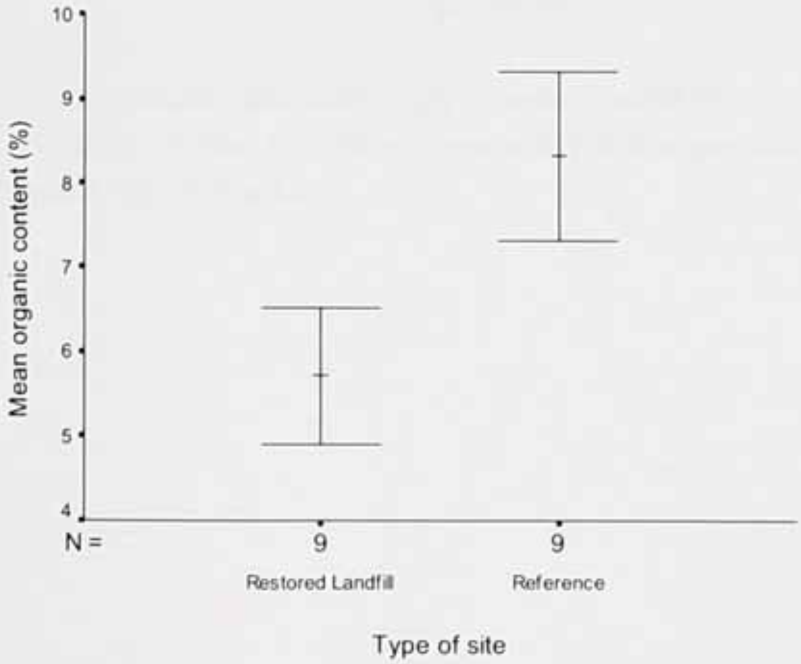


Figure 3.11 Mean organic content (%) for restored landfill and reference sites (±95% Confidence Limits). N=Sample sizes. Paired samples t-test (two-tailed); Landfill vs. Reference: t=-5.146, df=8, p=0.001.

Figure 3.12 Mean stone content per sample for restored landfill sites and reference sites. N=Sample sizes. Solid bar – medians, O – outliers, * - extremes. Paired samples t-test (two-tailed); Landfill vs. Reference: t=4.30, df=8, p=0.003.

Comparing richness of plants in flower with soil characteristics

There were almost no significant relationships found between the richness of plants in flower and the soil characteristics for the restored landfill and reference sites (Table 3.07). Soil texture and pH were not analysed as there was no difference between the two types of sites. The one significant correlation found was that of stone content and richness of plants in flower on reference sites. Data for 2008 was not tested as the sample size was too small. There were no non-linear relationships between the soil variables and richness of plants in flower apparent for the data.

Table 3.07 Spearman's rank correlation between annual richness of plants in flower and soil characteristics for 2007. (two-tailed, r = correlation coefficient, sample size n=9).

Site type	Soil variable	Correlation (r)	Probability
Restored landfill	Bulk density	0.44	0.24
	Moisture	-0.42	0.26
	Organic	-0.39	0.30
	Stone	0.12	0.76
Reference	Bulk density	-0.15	0.70
	Moisture	-0.53	0.14
	Organic	0.13	0.73
	Stone	0.81	0.008

How do restored landfill sites compare with the reference sites for the provision of resources for flower-visiting insects?

The restored landfill sites were assessed for their average floral cover (cm^2 per m^2) and their total area of floral cover on-site. At their flowering peak in 2007, the nine restored landfill sites surveyed, with a mean size of 12.8 ha., had a mean floral cover of 6.6 cm^2 per m^2 and a combined total floral cover of 643 m^2. This compared to the nine reference sites, with a mean size of 6.3 ha., having a mean floral cover of 10.1 cm^2 per m^2 and a combined total floral cover of 342 m^2, see also Table 3.08.

Table 3.08 Peak mean and total floral cover in 2007 a) restored landfill sites, b) reference sites.

Type	Site	Floral cover (cm^2 per m^2)	Size (ha.)	Total floral cover (m^2)
Restored	Kettering	22.07	10.80	238.33
	Harlestone	14.31	6.60	94.46
	Wootton	7.10	14.57	103.38
	Sidegate Lane	5.12	4.13	21.13
	Brogborough	4.68	26.11	122.28
	Cranford	2.81	0.58	1.63
	Brixworth	2.44	11.25	27.45
	Bletchley	1.02	34.00	34.77
	Kilsby	0.01	7.19	0.05
Reference	Scrub Fields	25.27	3.00	75.80
	Blue Lagoon	15.73	1.05	16.52
	Glebe Meadows	15.72	4.12	64.78
	Pitsford	12.6	0.79	9.96
	Draycote	7.90	2.25	17.78
	Twywell	6.84	8.08	55.26
	River Ise Meadows	3.65	21.36	77.88
	Barnes Meadows	1.57	4.18	6.56
	Ditchford	1.43	12.19	17.49

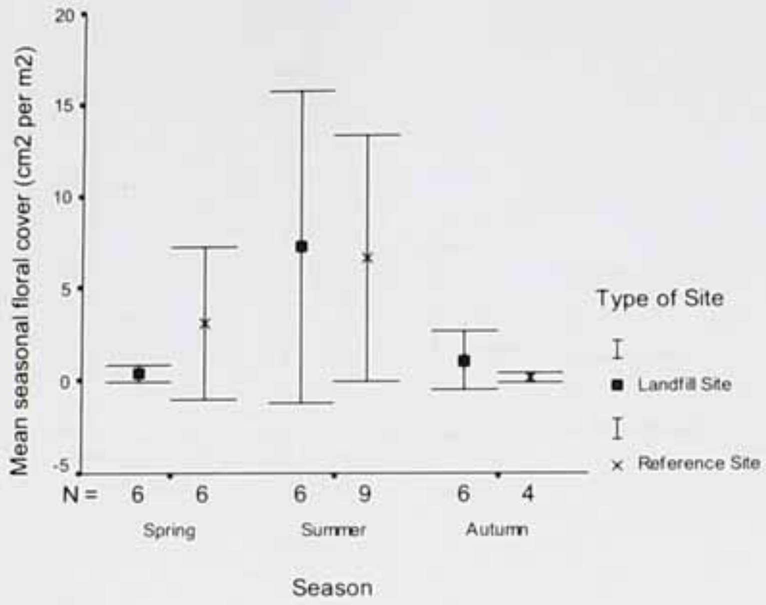
Seasonal floral cover on restored landfill and reference sites.

For 2007 there was a significant seasonal effect on the floral cover (i.e. cm^2 of floral bloom per m^2 of transect) for restored landfill sites but not for reference sites (Figure 3.13a). There was no significant difference between the floral cover of the restored landfill and reference sites for any season (Figure 3.13a). For 2008 there was no significant seasonal effect on the mean floral cover for either the restored landfill or for the reference sites (Figure 3.13b). There was no significant difference between the floral cover of the restored landfill and reference sites for any season (Figure 3.13b).

Seasonal total floral cover on restored landfill and reference sites.

For 2007 there was a significant seasonal effect on the total floral cover (i.e. m^2 of floral bloom per site) for restored landfill sites but not for reference sites (Figure 3.14a). There was no significant difference between the total floral cover of the restored landfill and reference sites for any season (Figure 3.14a). For 2008 there was no significant seasonal effect on the mean total floral cover for either the restored landfill or for the reference sites (Figure 3.14b). There was no significant difference between the total floral cover of the restored landfill and reference sites for any season (Figure 3.14b).

The mean annual totals were calculated for restored landfill and reference sites, this being the sum total of the mean seasonal values. For 2007, the reference sites had a greater accumulation, whilst in 2008 this was reversed (Figure 3.15).



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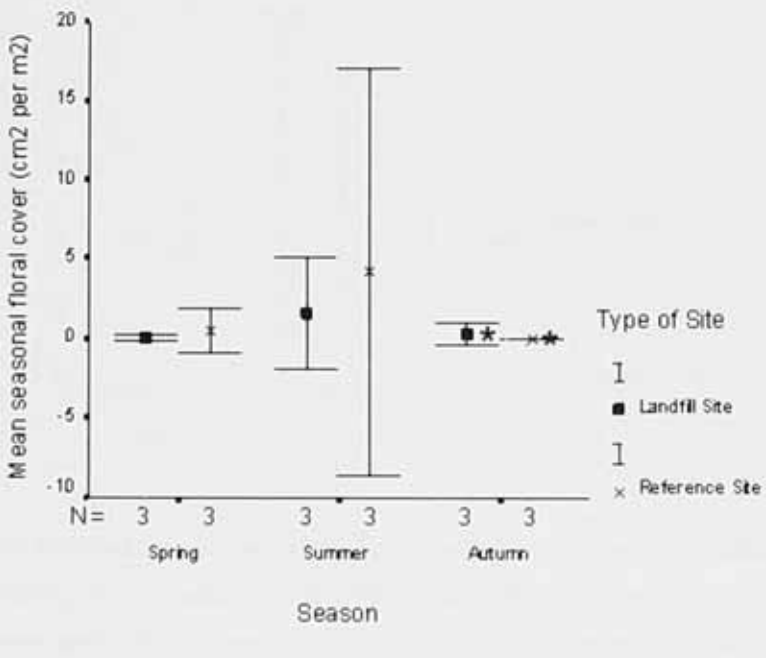
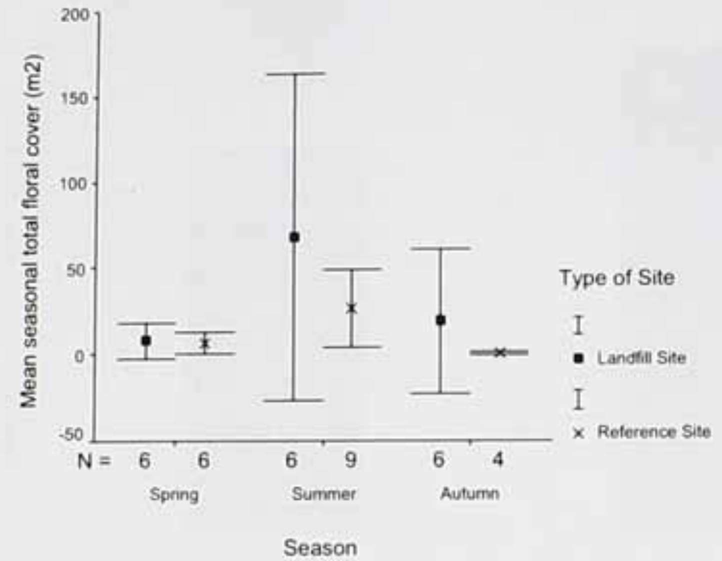


b)

Figure 3.13 Mean floral cover per season for restored landfill and reference sites (cm^2 per m^2). N=sample sizes. * - Medians a) 2007 One-way ANOVA; Restored landfill sites across seasons $F_{2,17}$=3.84, p=0.05. Reference sites across seasons $F_{2,18}$=1.49, p=0.26. Paired samples t-test (two-tailed) Spring Landfill vs. Reference t= -1.07 df=4 p=0.34, Summer Landfill vs. Reference t=0.29 df=5 p=0.79, Autumn Landfill vs. Reference t= 0.66 df=1 p=0.63. b) 2008 One-way ANOVA; Restored landfill sites across seasons $F_{2,8}$=2.77, p=0.14. Reference sites across seasons $F_{2,8}$=1.74, p=0.25. Paired samples t-test (two-tailed) Spring Landfill vs. Reference t= -1.23 df=2 p=0.34, Summer Landfill vs. Reference t=-1.01 df=2 p=0.42. Wilcoxon test (two-tailed): Autumn Landfill vs. Reference Z=-1.34 p=0.18.



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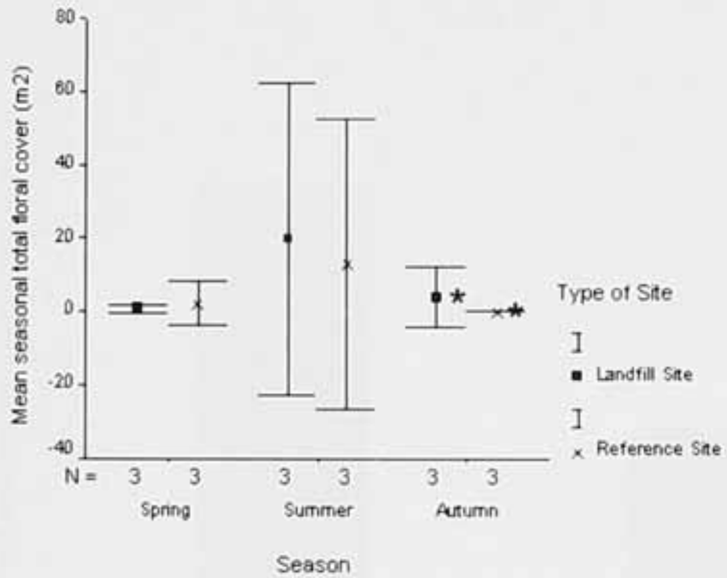
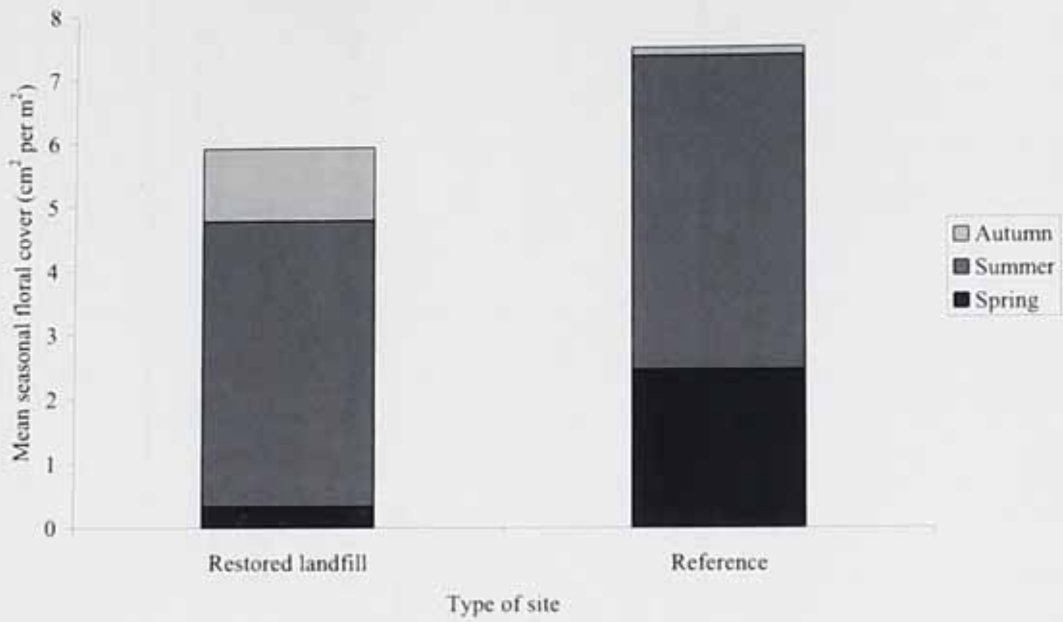


b)

Figure 3.14 Mean seasonal total floral cover per site for restored landfill and reference sites (m^2). N=sample sizes. * - Medians a) 2007, Restored landfill sites across seasons $F_{2,17}$=1.88, p=0.19. Reference sites across seasons $F_{2,18}$=2.88, p=0.09. Paired samples t-test (two-tailed) Spring Landfill vs. Reference t= 0.09 df=4 p=0.94, Summer Landfill vs. Reference t=1.17 df=5 p=0.30, Autumn Landfill vs. Reference t= 3.64 df=1 p=0.17. b) 2008 One-way ANOVA; Restored landfill sites across seasons $F_{2,8}$=3.09, p=0.12. Reference sites across seasons $F_{2,8}$=1.71, p=0.26. Paired samples t-test (two-tailed) Spring Landfill vs. Reference t= -0.88 df=2 p=0.47, Summer Landfill vs. Reference t=0.36df=2 p=0.75. Wilcoxon test (two-tailed): Autumn Landfill vs. Reference Z=-1.34 p=0.18.



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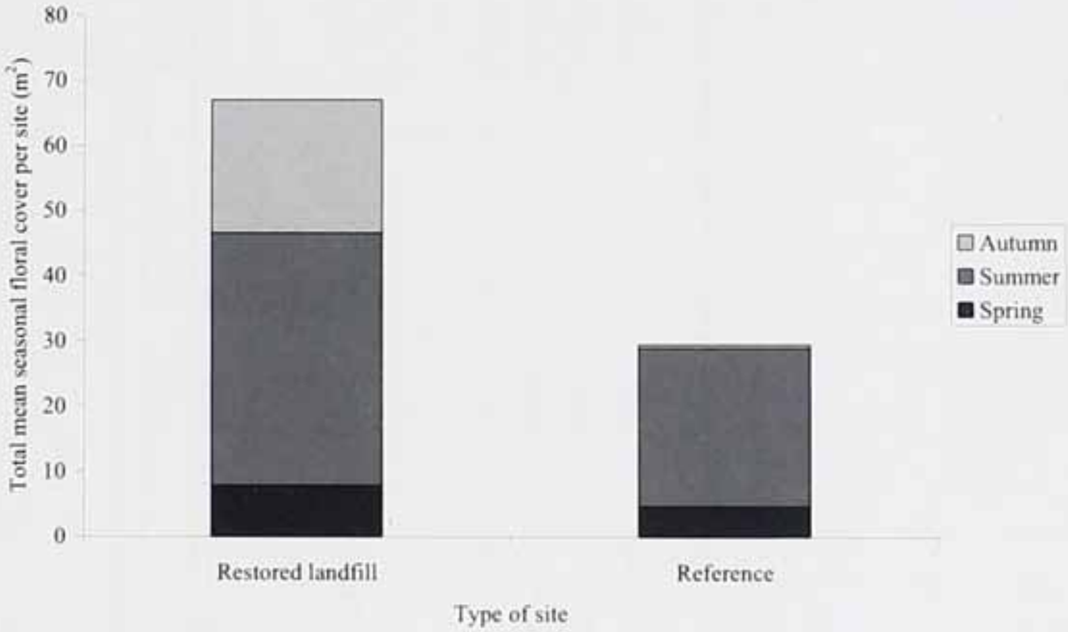


b)

Figure 3.15 Cumulative seasonal floral cover for restored landfill and reference sites 2007 a) mean seasonal floral cover (cm^2 per m^2) per site b) total on-site mean seasonal floral cover (m^2) per site.

The results for this section have been summarised in Table 3.09.

Table 3.09 Summary of floral resource provision on restored landfill sites and reference sites.

Temporal scale	Variable	Year	Significance	Result
Annual	Cumulative seasonal floral cover (cm^2 per m^2)	2007	na	Reference sites have greater cumulative mean seasonal floral cover, cm^2 per m^2 than restored sites
Annual	Cumulative total seasonal floral cover (m^2)	2007	na	Restored sites have greater cumulative mean seasonal cover per site m^2, than reference sites.
Seasonal	Mean floral cover (cm^2 per m^2)	2007	n.s	No difference.
Seasonal	Mean floral cover (cm^2 per m^2)	2008	n.s	No difference.
Seasonal	Total floral cover per site (m^2)	2007	n.s	No difference.
Seasonal	Total floral cover per site (m^2)	2008	n.s	No difference.

n.s. – not significant

Discussion

The aim of this chapter was first to consider whether restored landfill sites have been revegetated comparably with reference nature sites for their insect pollinated flowering plants and secondly to examine the resources available for flower-visiting insects.

The key finding for this chapter has been how floristically similar the restored landfill and reference sites are. The NMDS analysis shows that the majority of sites clump together regarding their flowering plant species and floral abundance (Figure 3.03). There is also great similarity regarding the plant families and their species richness (Table 3.04 and Figure 3.01). The most common and abundant species found on the restored landfill and reference sites are common grassland species (Tables 3.03 & 3.04).

It is significant that within this study approximately one quarter of plants are found exclusively on restored landfill sites, one quarter exclusively reference sites and one half shared on both (Figure 3.02). This shows that within the landscape and conservation context different types of sites, both older reference and newly restored sites are important for the preservation and conservation of plant species. The different types of sites are supporting and hence conserving different species of plants. It is unlikely that the species of insect pollinated plants will not be supported elsewhere within the landscape as they have dispersed to the sites. However, the more non-agricultural plants and floral resources there are within the landscape, the more resources there will be for associated herbivores, frugivores and flower-visitors. This may also illustrate the importance of perturbed habitats within the landscape. Historically these would have been more common, when the land was less tamed, and wild rivers were allowed to largely engineer the landscape, this availability of bare earth encouraged the growth of opportunistic plants (Kleem, 1996). Although bare earth is commonly turned over in the agricultural processes, opportunistic species are readily removed with herbicides on typical farm practice (Freemark and Boutin, 1995). These pioneer species may be important for the variety of animals that gain benefit from them such as seed eaters or herbivores.

How does richness of plants in flower and floral abundance compare between restored landfill sites and reference sites? How does richness of plants in flower and floral abundance compare between sown and naturally revegetated landfill sites?

The restored landfill sites and reference nature sites are similar in terms of richness of plants in flower and floral abundance when analysed for their annual means (Figures 3.04 & 3.06). This is surprising since there are perceived unfavourable conditions associated with restored landfill sites such as compacted, poorly laid soils and planted, grass-rich swards (Ettala et al., 1988; Wong, 1988; Davis and Coppeard, 1989; Lehmann and Rebele, 2002). Therefore, to find a relatively rich flowering plant community other ecological processes may be occurring. The restored landfill sites with their relatively young age and management regime of summer mowing may be maintained in a state of early succession, where there is generally a greater richness of pioneer plant species (Denslow, 1980). Conversely on the more established, older reference nature sites, later in successional stage, the vegetation will have fewer plant species, as there is little or no opportunism or perturbation to the system. However succession theory predicts highest species richness in mid-successional stages of "intermediate disturbance" (Connell, 1978; Brown and Southwood, 1987), therefore there is no significant difference between the two types of site.

The policies brought in to regulate the restoration and aftercare use of landfills have had the unforeseen effect of creating well vegetated, relatively florally diverse restored sites (Table 3.08). The seed mixes where used on those restored sites which are sown are cheap and grow readily on the soils, but may experience difficult environmental conditions such as drought in the summer, and water-logging in the winter (Gilbert and Anderson, 1998; Watson and Hack, 2000). Previously, research into plant diversity on landfill sites in the UK from the late 1980's and early 90's, have found relatively low species richness (Wong, 1988; Ireland, 1991). Reasons for this are attributed to the poor restoration standards and containment measures, where methane emissions through the soil affect plant growth. The current restoration practice on modern landfill sites and hence on those within this study has no issues of gas leakage, being contained within an engineered cap, and removed for fuelling electricity generation.

Restored landfill sites are relatively homogenous environments, having had graded soils and may be sown, so a lower diversity would be expected. The ryegrass seed mixes used by landfill restoration may actually act as a 'nurse-crop'. They provide early green cover which helps suppress weeds whilst the slower growing perennials establish (Hutchings et al., 2006). Over time these initial species will be eventually excluded by the perennials; however the nurse crop will itself slow the establishment (Mitchley et al., 1996). In comparing the naturally revegetated and sown restored landfill sites there was no difference in the floral abundance and plant richness in 2007 (Figures 3.07 & 3.08) . It does seem that natural revegetation of restored sites would be a cost-effective and efficient method for revegetating the sites. Empirical studies need to be undertaken to assess the specific benefits from both methods of revegetation.

When assessed seasonally for floral cover, there was the expected shift with an increase in the summer peak. There is however a noticeable difference between site types, with the reference sites having increased richness and abundance in the spring, and the restored landfill sites increased richness and abundance in the autumn (Figures 3.05, 3.06, 3.13 & 3.14). Reasons for this may be the management methods adopted on the sites as the reference sites are often mown in the early autumn. This could have important value with regards to abundance of floral resources within the landscape. The restored landfill sites maybe 'filling in' a hole in the floral resources on the landscape scale, particularly in our post-industrialised agricultural landscape (Figure 3.16). Sequential flowering on restored sites and across landscapes may maximise seed set of plant species and sustain populations of generalist pollinators (Waser and Real, 1979). Positive aspects of this for flower-visiting insects will be discussed in Chapter 4.

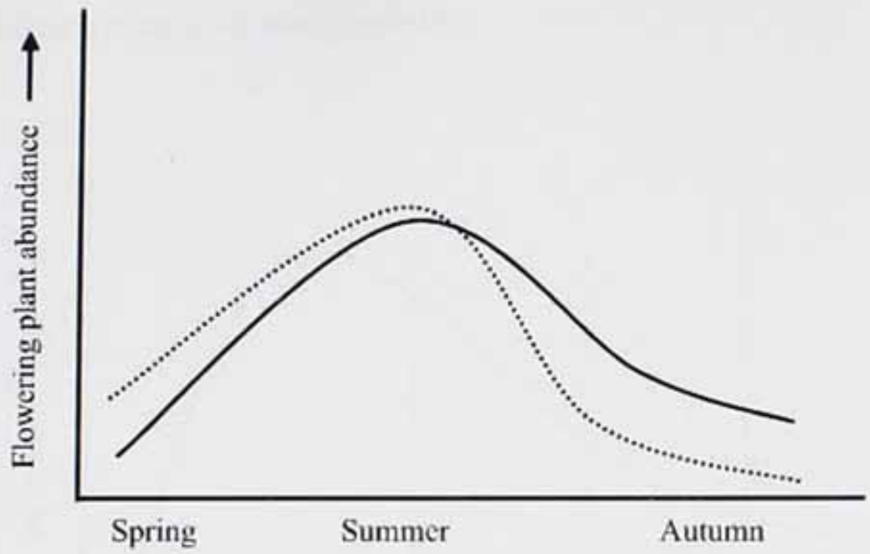


Figure 3.16 Schematic of floral resource provision in the landscape from restored landfill sites and reference nature sites. Solid line – floral resource available of restored landfill sites. Dashed line – floral resources available on reference nature sites.

What effect do soil characteristics have on species richness of plants in flower on restored landfill sites?

The fact that there was no underlying relationship between the soil characteristics and flowering plant richness, as expected, was surprising (Table 3.07). Within this study the soil quality was assessed, and restored landfills sites were shown to be poorer having greater compaction, drier, lower humus and stonier soil (Figures 3.09-3.12). That the restored landfill sites have similar floral characteristics to the reference sites even though they have poorer quality soil is interesting. It has been found that patterns of plant richness in relation to nutrient levels generally follow a hump-backed-curve; species richness being low at low nutrient levels, increasing to a peak an intermediate levels, and declines more gradually with further increasing nutrient levels (Pausas and Austin, 2001). The soil quality measured here may be following a similar trend. This may indicate that the restored sites had reached the stage of gradual decline post-peak richness of the relationship curve (Figure 3.17).

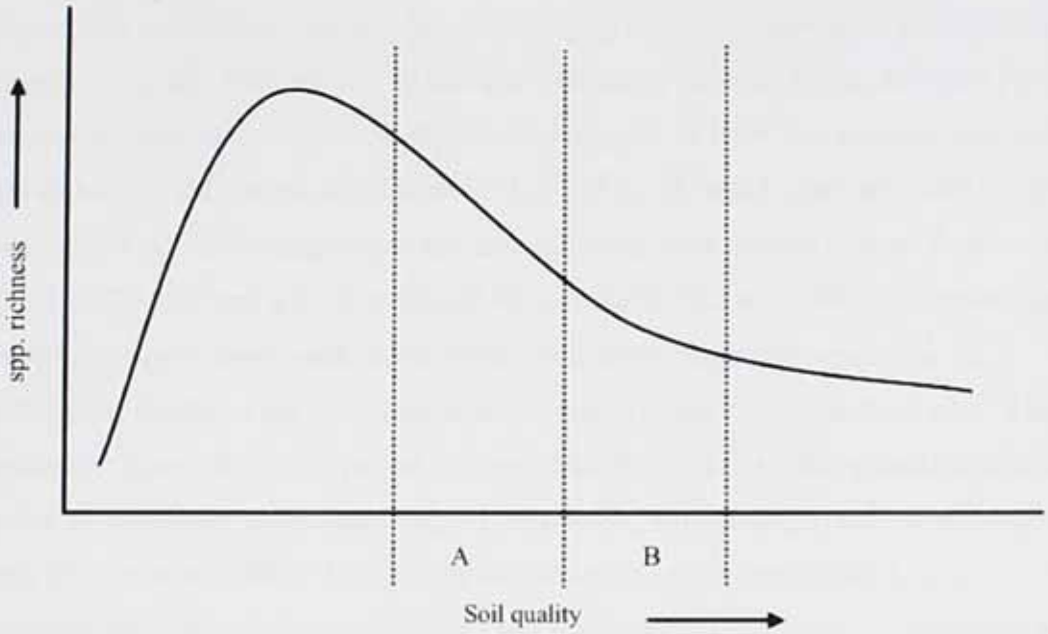


Figure 3.17 Theoretical curve of relationship between plant species richness and soil quality. A – possible location of landfill sites, B – possible location of reference sites.

The use of poorer quality soils within landfill restoration may increase richness of insect pollinated flowering plants, since they will not be out competed by grass species. The most promising sites for natural colonisation are those with poor soils (Gilbert and Anderson, 1998). Restoration methods which favour poor soils may generally enhance flowering plant diversity, an effect that has been shown for different taxa at the habitat level (Murdoch et al., 1972; Potts et al., 2003; Tylianakis et al., 2005). The Environment Agency promotes the use of higher quality soils as they envisage an agricultural future use (Environment Agency, 2004), however poorer quality soils are actually better for the flowering plant assemblage. Landfill site operators would also benefit from this as poorer soils are easier and cheaper to obtain for restoration than nutrient rich ones. Of clear issue is the need for more research with relation to the ecological benefit of different types of soils which could be used in landfill restoration. This aspect was not taken further as it was only intended to play a small measure within this study, and time and resources were limited.

How successful are restored landfill sites for the provision of resources for flower-visiting insects?

The restored landfill sites are providing similar floral resources as the reference sites (Figure 3.15 and Tables 3.08 & 3.09). Calculating the overall resources provided by the restored suites was difficult, due to being a continually variable figure, but their peak rate may be indicative of this. At their flowering peak in 2007, the nine restored landfill sites surveyed, had a mean floral cover of 6.6 cm^2 per m^2 and a combined total floral cover of 643 m^2. This compared to the nine reference sites, having a mean floral cover of 10.1 cm^2 per m^2 and a combined total floral cover of 342 m^2. Although the restored landfill sites had a lower peak mean floral cover, their larger size meant that on a landscape scale they were providing nearly twice as much as the reference sites. The floral cover mean of 6.6 cm^2 per m^2 is lower than that found in other grassland research, studies of grasslands in Germany ranged from 5-20.5% (500-2050 cm^2 per m^2) floral cover (Meyer et al., 2009). Reasons for this however may be attributed to their interpretation of the method, as it is not clear if they have used flower or flowering plant cover. There was a seasonal difference observed with a greater floral cover for the restored landfill sites than the reference sites in the autumn (Figures 3.13 & 3.14). This again is indicating the important value of restored landfill sites within the landscape as described previously (Figure 3.16).

In conclusion, the restored landfill sites inadvertently provide natural floral resources, even when this has not been their intended restoration outcome. They support a species rich and florally abundant plant community as do the reference nature sites. Their floral resources help enrich the ecological landscape, particularly when other resources have diminished. The landfill sites are cut in the summer to reduce the associated fire risk and left alone later into the year, providing resources when they maybe absent elsewhere when nature sites are cut or grazed. The area of land under landfill use in the UK means that their restoration has the potential to significantly benefit native flowering plants and their supported wildlife.

Results summary

In conclusion, the results of this part of the study can be summarised as follows:-

- Approximately one quarter of species unique to both restored landfill sites and reference sites, and one half found on both types.
- No differences were found in plant richness and floral abundance between restored landfill sites and reference sites.
- Seasonal variation in the floral richness and abundance was apparent. In the spring restored landfill sites had lower species richness and floral abundance. In the autumn they had higher species richness and floral abundance.
- No difference was found in floral characteristics relating to revegetation method, either seeded or naturally revegetated landfill sites.
- There was a significant difference found in soil properties between restored landfill sites and reference sites, but no correlation with species richness on the restored sites.
- No difference was found in the cumulative floral cover between restored landfill and reference site.

Figure 3.18 A typical restored landfill site – Brixworth June 2008

Chapter

4

A comparison of flower-visiting insects on restored landfill and reference sites.

A comparison of flower-visiting insects on restored landfill and reference sites.

"When one tugs at a single thing in nature, he finds it attached to the rest of the world."
John Muir (1838-1914), Naturalist, preservationist and founder of the Sierra Club

"A hypothesis or theory is clear, decisive, and positive, but it is believed by no one but the man who created it. Experimental findings, on the other hand, are messy, inexact things, which are believed by everyone except the man who did that work."
Harlow Shapley (1885 – 1972)

Summary

Flower-visiting insects have seen a decline in suitable habitat in the UK (Osborne and Corbet, 1994), and restored landfill sites may be a valuable habitat for supporting flower visiting insects. This chapter describes the plant-flower visitor community found on restored landfill sites in relation to the success of restoration. Plants in flower and their flower-visiting insects were compared on nine pairs of restored landfill and reference nature sites in the first year and three pairs of sites in the second year. The sites were surveyed through spring, summer and autumn to sample the complete flowering phenology. Flower-visiting insects were surveyed using belt transects; each site was surveyed three times during each fieldwork day to sample for early, mid and late resource gathering.

A total of nearly 1000 insects were collected over the two field seasons. Approximately one quarter of species were unique to restored landfill sites, one quarter to reference sites and one half were shared on both types of sites. The restored landfill and reference sites had similar flower-visiting insects with regards to abundance and species richness. A positive relationship was found between flower visitor abundance and seasonal floral cover, and richness of flowering plants and flower visitors, for both restored landfill and reference sites. The habitat quality analysis showed that a few flower-visiting insect specific features, such as bare earth and suitable holes, had an effect on the flower

visitor richness and abundance but that the most significant effect came from the flowering plant species richness. Overall, the restored landfill sites were similar to the reference sites in supporting an abundant and rich assemblage of flower-visiting insects.

Introduction

The conservation of flower-visiting insects is important given their current decline (Williams, 1982; Allen-Wardell et al., 1998; Goulson et al., 2002; Goulson et al., 2005; Biesmeijer et al., 2006; Stout, 2007; Colla and Packer, 2008; Gallai et al., 2009; Potts et al., 2009). Species of flower-visiting insects are important both in their own right, but also for the ecosystem services they provide (for further information relating the state of flower-visiting insects, their conservation and decline see Chapter 1). It is desirable that for successful ecological restoration, all biological services must be reinstated. Restoration will only be successful when the functional processes and interactions are re-established. This is developed and assessed in Chapter 5, relating to the interaction webs on the sites. Restored landfill sites may be a valuable resource of land for aiding the conservation effort for flower-visiting insects. The restoration of landfill sites is not currently targeted towards wildlife habitats but incidentally may be supporting flower-visiting insects (for further information on the restoration of landfill sites see Chapter 1).

Chapter 3 showed that the restored landfill sites were similar in their floristic characteristics to the reference sites, in their species richness and floral abundance. The restored landfill and reference sites are both grassland habitats; which are known to support rich communities of flower-visiting insects (Steffan-Dewenter and Tscharntke, 2002; Ebeling et al., 2008; Franzén and Nilsson, 2008; Noordijk et al., 2009; Potts et al., 2009). Flower-rich grasslands provide floral resources whose association with flower-visiting insects has been demonstrated (Smart et al., 2000; Carvell, 2002; Potts et al., 2003a). The floral resources available on the restored sites have been assessed (Chapter 3), and this study now examines the success of landfill restoration for supporting flower-visiting insects. 'Restoration success' in this context refers to the restored landfill sites having a similar level of species richness and abundance of individuals to the reference sites.

Flower-visiting insects have habitat requirements that must be met before populations will become established in restored communities. For example, bees require both nectar and pollen food resources, nesting sites and nesting materials (Westrich, 1996; Kearns et al., 1998). This study will focus on habitat quality in terms of floral resources, and other physical environmental factors (e.g. nesting sites and material, south facing slopes, vegetation structure etc). 'Floral resources' refers to the both the floral abundance and richness of insect pollinated plants in flower (See Chapter 3).

Aim

The aim of this chapter is to determine how successful restored landfill sites are in supporting flower-visiting insects, with regards to individual species and groups. This aim will be addressed in the following research questions:

What species and groups of flower-visiting insects are found on restored landfill sites? How does the flower visitor abundance and species richness compare between restored landfill and reference nature sites?

As it has been shown in Chapter 3, the restored landfill sites and reference sites are similar in terms of their floral characteristics. The expectation would be that the restored sites may have a lower flower visitor species richness and abundance than the reference sites. This relates to the dispersal and colonisation abilities of the insects, and some species of flower-visiting insect may not have colonised the restored landfill sites yet. The restored sites which are older therefore may have a greater subset of insect species, due to those insects with poorer dispersal abilities have had longer to colonise from the landscape's species pool. Therefore regarding flower visitor species richness, it is expected that there will be fewer flower-visiting insect species on the restored sites, and it is predicted that those older sites may have the greatest species richness. It would be expected that those sites with the greatest floral abundance will also have the greatest flower-visiting insect abundance.

How does floral resource use compare between landfill and reference sites?

The relationship between flower visitors and floral resources will be further examined. The 'floral resource use' in this context refers to the tightness of this relationship. Positive relationships have been found between the richness of flowering plants and abundance of floral resources and the diversity of their flower visitors (Lagerlof et al., 1992; Steffan-Dewenter and Tscharntke, 1999; Backman and Tiainen, 2002; Carvell, 2002; Steffan-Dewenter et al., 2002; Klein et al., 2003; Potts et al., 2003a; Westphal et al., 2003; Potts et al., 2004; Maccherini et al., 2009). A weaker relationship between the flower visitors and floral resources on restored landfill sites may mean that the flower visitors are using the surrounding landscape to a greater extent.

How do habitat quality features determine the flower visitor assemblage? How do these features compare between restored landfill sites and reference sites?

This question will be addressed by comparing the habitat quality features between restored landfill sites and reference sites, and correlating them against flower visitor richness, abundance and how they correlate with different groups of insects.

The restored landfill sites are newly created, relatively homogeneous environments which may lack habitat quality features which benefit flower visitor species and groups, such as dead vegetation and suitable abandoned rodent holes. In contrast, reference sites have been managed for their ecological benefit, and have relatively natural ecosystem dynamics occurring. It has been shown that high plant or structural resource diversity can correlate with high insect diversity (Murdoch et al., 1972; Fitter, 1982; Benton et al., 2003; Lundholm and Larson, 2003; Potts et al., 2003a; Baer et al., 2005; McMaster, 2005). The expectation would therefore be that those sites determined to have poor habitat features have consequently have poorer flower visitor assemblages, and that different flower-visiting insect groups may react differently to the habitat quality variables.

Methods

Study region and study sites

The study was conducted in the East Midlands of the UK, in the counties of Northamptonshire, Bedfordshire, Warwickshire and Buckinghamshire. All of the sites were within 50 km of Northampton. Nine landfill and paired reference sites were surveyed in 2007 (See Chapter 2 - Table 2.01). In 2008, four landfill and paired reference sites accompanied by paired reference sites were initially surveyed, although this was reduced to three due to logistical issues. For further details on site selection, see Chapter 2 – Methods.

Fieldwork timing

Fieldwork surveys were conducted from March to October 2007 and 2008, as this corresponds to the main flowering period in central England and hence the flower visitor activity. Local weather conditions made uniform distribution of sampling days impossible. For distribution of survey days see Chapter 2 - Table 2.02.

Flower visitor surveys

Flower visitor surveys were undertaken three times between 9am and 4pm on days which were warm and sunny with little or no wind, as outlined in the Butterfly Monitoring Scheme (Pollard and Yates, 1993) and similar to those used in previous pollination studies (e.g. Goverde et al., 2002; Kleijn and van Langevelde, 2006; Potts et al., 2006; Nielsen and Bascompte, 2007). Surveys each lasted 30 minutes and all flower visiting insects seen to be feeding legitimately (i.e. not nectar robbing) and large enough to touch anthers and stigmas were captured.

In the first year of study, 2007, plants in flower and flower visitors were surveyed using belt transects. This method has often been used in this type of research, being the most useful in estimating the number of insects across a range of sites (Banaszak, 1980; Kearns and Inouye, 1993; Dicks et al., 2002; Dafni et al., 2005; Forup and Memmott, 2005). The plants in flower, as recorded in the floral surveys, refer to those plants which are insect pollinated i.e. not grasses or other wind pollinated species. Flower visitors were surveyed along the transect line as defined in the floral survey section (See Chapter 3 - Methods). The transect was left undisturbed for 20 minutes following the initial entomophilous plant survey to allow the flower visitors to return. The 100m x 2m transect was surveyed at a rate of approximately 3 metres per minute for 30 minutes. Insects were captured using a butterfly net and placed into individually labelled specimen jars. After the survey was completed, those insects that could be identified in the field were recorded and released. Any time spent in transferring captured insects to jars was deducted from the total time to achieve a constant sampling effort. No distinction was made between different types of feeding behaviour i.e. pollen versus nectar as it is not always easy to distinguish between them in such surveys. The first species of plant which the insect was seen visiting was the one recorded. The flower visitor survey was conducted on transects at three different times (the morning, within one hour of midday and the afternoon) to sample those insects active at different times.

Those insects that could not be identified in the field were collected as voucher specimens for later identification. The specimens were identified using specific guides and/or keys: hoverflies (Stubbs and Falk, 2000), bumblebees (Prys-Jones and Corbet, 1991), beetles (Chinery, 2005), flies (Colyer and Hammond, 1951; Erzinclioglu, 1996), bees (Michener, 2007), wasps (Zahradnik and Severa, 2000) and butterflies (Tomlinson and Still, 2002). The identification was verified by reference to collections and natural history special interest groups e.g. Northamptonshire Natural History Society Diptera study group.

In 2008 the surveying followed a spiral from a randomly determined point on the restored landfill sites, at a standard pace of 10 metres per minute for 30 minutes. This was similar to the survey method used by Nielsen and Bascompte (2007) and determined to be effective for ecological surveys (Kalikhman, 2007). Given a two metre wide transect, an area of approximately 600m^2 was sampled in each survey. This method allowed for a greater area to be surveyed, which was required due to the relatively low flower-visiting insect density.

Habitat quality features

The local environmental variables contributing to habitat quality were assessed namely: area, and vegetation structure; suitable nesting sites including: bare earth, sandy soil, microstructures, south facing slopes, and hedgerows and shrubs; and floral resources relating to flowering plant richness and floral abundance [Table 4.01 and Figures 4.01 a)-l)]. The search for general characteristics which may be used to define the habitat quality for flower-visiting insects is confounded by the fact they are relatively species specific. However, they do give a measurable variable given that most flower-visiting insect species' life histories and specific plant requirements are not known. The ordinal scale (1-5) used here to assess habitat variables is: 1 having few or a low abundance, whilst 5 having many or high abundance. This scale is appropriate and useful given that all observations were made by one observer and therefore data is fair across all sites. This method of recording habitat variables has been used in previous studies (Sjödin et al., 2008).

Vegetation height and density

A rising plate meter was used to estimate the vegetation height and density (Earle and McGowan, 1979; Sharrow, 1984). The sward canopy height and forage bulk were measured using a metre stick inserted in the centre of the plate. Canopy height was determined by lowering the disc until the upper most leaves were in contact with the plate. The forage bulk was measured by dropping the disc from the top of the metre stick onto the vegetation and then reading off from the measure.

Complete on-site plant species surveys

Plant species were recorded from within 20 x $1m^2$ quadrats laid regularly along two 100m transects perpendicular to each other, crossing at the approximate centre of the site. All plants were identified to species level. Further details of plant surveys are available in Rahman (2009).

Other measured variables methods and rationale are discussed in Table 4.01.

Table 4.01 Description of environmental variables, methods and rationale, attributing to habitat quality.

Variable	Description and Method	Rationale
Area of habitat site	Area of habitat site, using GIS software and Ordnance Survey data.	Size of the site as an area of semi-natural vegetation.
Vegetation structure	Mean of 20 measurements using a rising plate meter (Sharrow, 1984; Sanderson et al., 2001). Both maximum height and density.	Indicative of management intensity (Sjödin et al., 2008).
Suitable nesting earth	Area presence of bare loose earth or sand soil. 1-5 scale.	For availability of nest building material and sites.
Microstructures	Area presence of distinct features within the ground surface, including holes, cracks, tussocks, visual stones (Morris, 1969). 1-5 scale.	Micro-heterogeneity of habitat site.
South facing slopes	Abundance of south facing slopes (Figure 4.01 b). 1-5 scale.	Presence of suitable nesting south facing slopes.
Hedgerows and shrubs	Presence of hedgerows and shrubs within or bordering the habitat site. 1-5 scale.	Heterogeneity of habitat.
Flowering plant richness	Number of flowering plant species.	Diverse resource availability.
Total plant species richness	Number of plant species, using 10 quadrats placed on-site within a diagonal arrangement. Identifying all species within.	Diverse plant resource available for alternative plant uses – e.g. larval hosts (Krauss et al., 2003).
On-site flower abundance	Counts of flowers per metre squared. For definition of floral unit see Chapter 2.	Nectar resource available to flower-visiting insects.

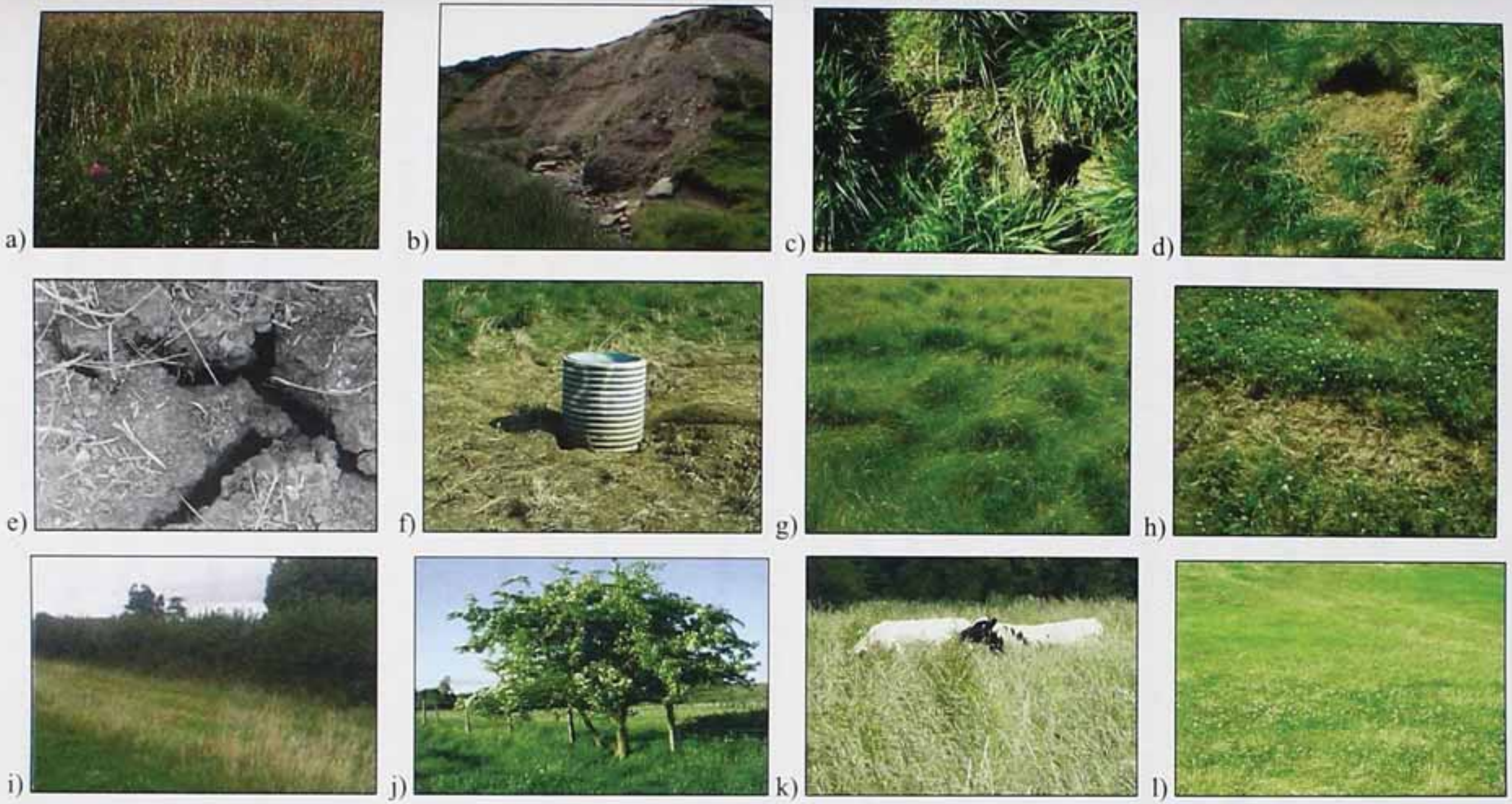


Figure 4.01 Examples of habitat quality assessment features: **a) ant mound, b) suitable south facing slope, c) small rodent holes, d) rabbit diggings with loose bare earth, e) cracks in soil, f) landfill gas vent point, g) grass tussocks, h) area of lying dead vegetation, i) hedgerows, j) shrubs, k) tall vegetation and l) recently mown grass.**

Data analysis

The data were tested for normal distributions using one-sample Kolmogorov-Smirnov tests. Levene's test was used to determine whether variances were significantly homogenous, and, if heterogeneous, the significance levels were adjusted accordingly. For testing differences between the types of sites within the pairing, paired samples t-tests and Wilcoxon signed rank tests were used to compare parametric and non-parametric data, respectively. For tests of differences in three independent samples, one-way ANOVAs or Kruskal-Wallis tests were used for parametric and non-transformable non-parametric data, respectively; post-hoc tests using paired samples t-tests were then used. For testing of difference in presence and frequency of species between types of sites matched t-tests and Wilcoxon signed rank tests were used to compare parametric and non-parametric data, respectively. Correlation analysis has been undertaken using Pearson's correlation for parametric data and Spearman's rank correlation for non-parametric data. SPSS version 11.5 statistical software was used (SPSS, 2003). Abbreviations have been used: restored landfill and reference comparison sites may be referred to as 'landfill' and 'reference' respectively. Significant results: $p \leq 0.05$.

What species and groups of flower-visiting insects are found on restored landfill sites? How does the flower visitor abundance and species richness compare between restored landfill and reference sites?

Flower visitor species comparison

The presence and site frequency of all of the flower visitor insect species was determined for restored landfill and reference sites. Scatter plots were generated to illustrate the relationships between species present and site types. Species abundance and frequencies were ranked with relation to the reference sites to allow individual comparisons. Through assessing the data for both years judgment can be made towards which groups of flower visitors may be better suited to restored landfill or reference sites. This does relate to species within the groups having similar life history traits, which although questionable, could give an indication as to which sites have the most suitable habitat features for those groups of flower visitors. The frequency presence of groups was shown, which refers to the number of restored and reference sites the groups were found on.

Species composition and abundance between types of sites were represented using non-metric multidimensional scaling (NMDS). Intervals in the data were measured using Euclidean distance. Euclidean distance was used rather than Bray-Curtis as it gives a greater distortion and hence visual spread to the data, which is an advantage when the sites being compared are expected to be very similar (Kessell and Whittaker, 1976). Unlike other ordination techniques NMDS does not make assumptions as to or about the distribution of the variables (Maarel, 2005). NMDS instead uses rank distances for ordination, and so this gives a visual representation with those sites having similar composition closer together (Legendre and Legendre, 1998; Maarel, 2005; Ollerton et al., 2009). For further information regarding use of NMDS see Maarel (2005).

Assessments were made for differences in the abundance and species richness of flower visitors between restored landfill and reference sites, the significance of the seasonal effect, and between the site types within-season. There was no effect of habitat size on the flowering plant richness or floral abundance as tests showed no significant correlations (Pearson's correlation; restored landfill sites 2007: site size & plant richness $r = 0.44$, $p=0.72$, site size & floral abundance $r= -0.12$ $p=0.34$, reference sites: site size & plant richness $r = -0.16$, $p=0.25$, site size & floral abundance $r= -0.15$ $p=0.26$).

How does floral resource use compare between landfill and reference sites?

The blooming area or floral cover method used in this study combines floral abundance with inflorescence size (See Chapter 3). A positive relationship between nectar production and inflorescence size and number has previously been established (Harder and Cruzan, 1990; Holl, 1995; Pacini et al., 2003). Inflorescence number has been used as an indicator of sugar content (Sharp et al., 1974; Kremen, 1992; Munguira and Thomas, 1992; Forup and Memmott, 2005; Forup et al., 2008), and corolla size is less susceptible to plant vigour and hence site specificity than nectar secretion rates (Harder and Cruzan, 1990). Blooming area is sufficiently robust to allow comparisons across sites and has been used in previous pollination restoration studies (Steffan-Dewenter and Tscharntke, 2001; Carvell et al., 2004; Potts et al., 2006; Vulliamy et al., 2006; Clough et al., 2007; Holzschuh et al., 2007; Ebeling et al., 2008; Meyer et al., 2009).

The floral transect data for each site are presented as the mean within a season and compared against each of the flower visitor surveys undertaken in the same season. This is not pseudo-replication as the mean of the seasonal data represents the on-site floral resource at the time of the flower visitor transects, and was required owing to flower visitor surveys being taken across the whole site and not just contained within the floral transect area. Firstly, flower visitor abundance was compared with total mean on-site seasonal floral cover, and secondly, flower visitor species richness was compared with species richness of plants in flower. Richness of plants in flower refers to those species which are insect-pollinated, i.e. not grasses.

How do habitat quality features determine the flower visitor assemblage? How do these features compare between restored landfill sites and reference sites?

Correlations between flower-visiting insect richness and abundance against habitat quality features were done separately for restored landfill and reference sites. Only 2007 data were used as the 2008 sample was too small for this statistical analysis. Species richness for the separate groups of flower-visiting insects were correlated against the habitat quality features, for both restored landfill sites and reference nature sites.

Pearson's correlation was acceptable to use with the variable data, since although it is ordinal data, equal spacing of intervals is assumed. Bonferroni correction was used and those with a probability value below 0.005 were deemed as significant. This reduces type I errors but does increase type II errors, and so less stringent probabilities will also be examined.

Results

A total of 201 flower visitor surveys were performed, 129 in the first year on 18 sites and 72 in the second year on 6 sites. Over the two field seasons there were 942 flower-visiting insect samples taken, 317 the first year and 625 the second. This increase was due to the revised sampling method used. In the first year, the restored landfill sites yielded 156 individuals from 30 species of flower-visiting insects; the reference sites yielded 161 individuals from 37 species of flower-visiting insects. In the second year, the restored landfill sites yielded 405 individuals from 41 insect species and the reference sites yielded 220 individuals from 40 insect species. For the distribution of species between taxonomic groups see Table 4.02 & Figure 4.02. The groups are the main categories used in pollination studies and allow for comparison; no conclusions can be drawn from this for "functionality" of groups *sensu* Fenster et al. (2004); see also Ollerton et al. (2007). Species richness of groups of flower-visiting insects found on restored landfill and reference sites were analysed for significant difference using Fisher's exact test. No group species richness was significantly different between sites (Table 4.02).

Table 4.02 Species richness for groups of flower-visiting insects recorded on restored landfill and reference sites for 2007 and 2008. RL = Restored landfill, RF = Reference. Bees = all non-bumblebee bees, Bumblebees = Bombus and Psithyrus, and Butterflies = all butterflies and moths, Flies = all non-syrphidae, Other = all other flower-visiting insects. P= probability of significance using Fishers exact test.

		2007		2008		
Groups		**RL**	**RF**	**RL**	**RF**	**P**
Bees	(Hymenoptera \ Apoidea)	2	5	6	2	0.13
Beetles	(Coleoptera)	2	3	3	3	1.00
Bumblebees	(Hymenoptera \ Bombus)	5	4	4	5	1.00
Butterflies	(Lepidoptera)	1	6	6	8	0.34
Flies	(Diptera)	6	8	4	8	0.70
Hoverflies	(Diptera \ Syrphidae)	11	10	16	12	0.78
Other		3	1	2	2	1.00
Total		30	37	41	40	

The result from the Fisher's exact test analysis is surprising, given that some groups clearly have greater species richness in both years on one type of site. This lack of difference between site types for species richness of groups may therefore be a facet of the statistical test given the small frequencies used. The groups which have the same trend for both years between sites are more hoverfly species on the restored landfill sites, and more flies and butterflies species on the reference sites. The species distribution of butterflies has been further explored below (Table 4.03).

The species distribution of butterflies has been determined and their larval host species (Table 4.03). Whilst they share five species found on both types of site, uniquely two species are found on restored landfills and three on the reference sites. They generally share the same type of larval hosts on both types of site.

Table 4.03 Butterfly species present on restored landfill and reference sites for both years and their larval host species (Host species from Chinery 2005).

Site type	Butterfly species	Larval host plants
Restored landfill	*Aglais urticae**	Stinging nettle
	Maniola jurtina	Fine grasses
	Ochlodes sylvanus	Various grasses
	Pieris brassicae	Brassicas and nasturtiums
	*Polyommatus icarus**	Trefoil and clover
	Pyronia tithonus	Various grasses
	Zygaena filipenulae	Birds-foot trefoil
Reference	*Coenonympha pamphilus**	Fine grasses
	Maniola jurtina	Fine grasses
	*Melanargia galathea**	Red-fescue grass
	Ochlodes sylvanus	Various grasses
	*Panemeria tenebrata**	Mouse-ears
	Pieris brassicae	Brassicas and nasturtiums
	Pyronia tithonus	Various grasses
	Zygaena filipenulae	Birds-foot trefoil

* Species found uniquely on type of site.

Figure 4.02 a) and b) shows the species richness of flower visitor groups on the restored landfill and reference sites for 2007 and 2008, respectively. Hoverflies were found with greater species richness on restored sites, whilst butterflies and flies had greater species richness on reference sites. Other groups were less consistent; bees had greater richness on the reference sites in 2007, but a greater richness on restored landfills in 2008. It is unclear whether these shifts are related to an increased sampling effort, the reduced sub-set of sites or just random variation. Beetles and bumblebees remain relatively equally rich on both types of sites in both years. With the exception of the flies, all groups of flower-visiting insects increase in richness with the increased sampling of 2008.

Figures 4.02 c) and d) shows the abundance of individuals within the flower visitor groups. With the exception of bees, those groups which were found with the greatest abundance in 2007 also had the greatest abundance in 2008. The flower-visiting insect groups approximately split into two clusters, the bumblebees, hoverflies and flies with the greatest abundance and the butterflies, beetles and other groups with the lowest abundance. Only two groups were consistent in both years, with hoverflies and beetles most abundant on the restored landfill sites. Interestingly, all groups were more abundant on the restored landfill sites in 2008.

Figures 4.02 e) and f) shows the frequency with which groups were found on the restored landfill and reference sites, i.e. the number of sites on which a particular taxon occurred. For 2007, with the exception of butterflies all groups have a similar frequency on both types of sites. For 2008, bees and flies were found less on the restored sites, whilst all other groups are found with the same frequency on both kinds of site.

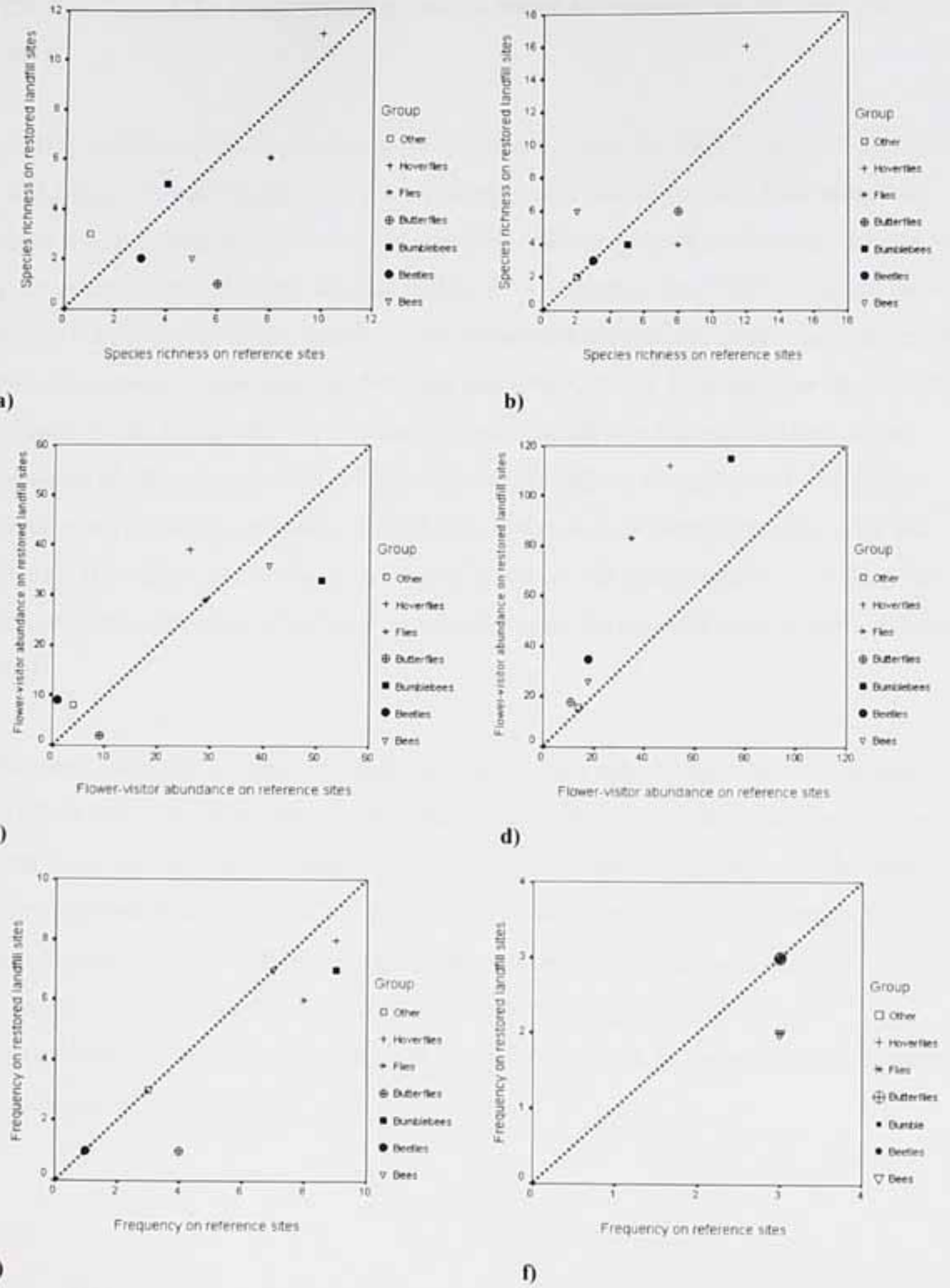


Figure 4.02 Comparison of flower visitor groups between pairs of restored landfill sites and their associated reference sites for: Species richness of groups a) 2007, b) 2008; Abundance of individual flower-visiting insects within groups c) 2007, d) 2008; Frequency of occurrence of groups on sites e)2007, f) 2008. NB. Freq. of occurrence of groups on sites for 2008 was: Bees and flies (3,2) all other groups at (3,3). Line = 1:1. Not all scales are the same.

Comparison of the flower-visiting species found on restored landfill and reference sites

NMDS ordination shows the similarity between the sites for 2007 (Figure 4.03). Eight of the nine restored landfill sites were placed closely within the plot, indicating they share many of their flower visitor species. The reference sites have greater variance in their spread, indicating they are less similar to one another. The NMDS S-stress value below 0.2 (S=0.142) is low and therefore shows that the data is represented truly by the two dimensional representation (McCune and Grace, 2002). Examples for the outliers include: Scrub Fields (H), the highest abundance of *Apis mellifera*, and the unique presence of *Helophus pendulus*; Kettering (6), the highest abundance of *Meligethes aeneus* and the unique presence of *Muscina prolapsa*; and Barnes Meadow with the highest abundance of *Bombus pascuorum*. However, the most significant finding from this ordination analysis is the large overlap between the restored landfill and reference sites.

The most abundant flower-visiting insect species for each site have been identified (Table 4.04). The most common abundant species for the restored landfill sites across both years, are the bumblebees, *B. terrestris/lucorum* and *B. lapidarious*. The other common species are flies and hoverflies. The most common abundant species for the reference sites across both years, are the bumblebees, *B. pascuorum*, *B. terrestris/lucorum* and *B. lapidarious* and the fly *Calliopum* spp. (Dipterta \ Lauxaniidae). Both the restored landfill and reference sites therefore share a number of their most abundant species.

Table 4.04 Most abundant flower-visiting insect species on restored landfill and reference sites. Abundance: total number recorded for all sites of that type. Site frequency: the number of sites present on.

Year	Site type	Insect species	Abundance	Site frequency
2007	**Restored landfill**	*Apis mellifera*	34	6 of 9
		Calliopum spp.	19	6 of 9
		Bombus terrestris/lucorum	12	6 of 9
		Bombus pascuorum	10	5 of 9
		Bombus lapidarius	10	4 of 9
	Reference	*Apis mellifera*	33	5 of 9
		Bombus pascuorum	26	6 of 9
		Calliopum spp.	17	5 of 9
		Bombus lapidarius	15	8 of 9
		Bombus terrestris/lucorum	6	4 of 9
2008	**Restored landfill**	*Calliopum* spp.	77	2 of 3
		Bombus terrestris/lucorum	49	3 of 3
		Bombus lapidarious	43	3 of 3
		Episyrphus balteatus	33	2 of 3
		Oedemera nobilis	27	3 of 3
	Reference	*Bombus lapidarious*	26	3 of 3
		Bombus pascuorum	25	3 of 3
		Calliopum spp.	24	3 of 3
		Bombus terrestris/lucorum	17	3 of 3
		Macropis europaea	16	3 of 3

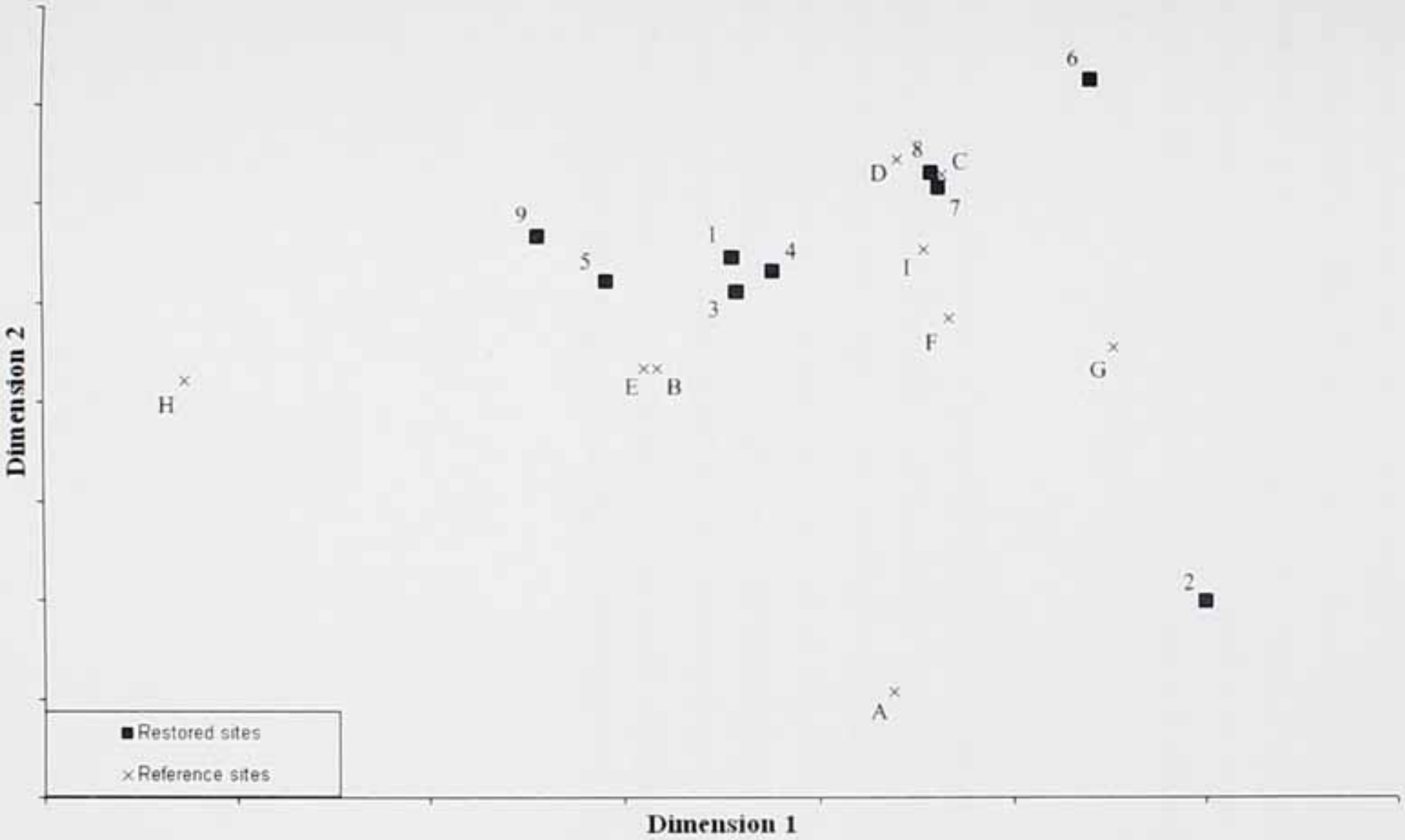


Figure 4.03 NMDS Ordination plot of flower-visitor species and abundance on restored landfill sites (1-9) and reference sites (A-I) for 2007. Two-dimensions used, S-stress = 0.142. Restored landfill sites: 1. Bletchley, 2. Brixworth, 3. Brogborough, 4. Cranford, 5. Harlestone, 6. Kettering, 7. Kilsby, 8. Sidegate Lane, 9. Wootton. Reference sites: A. Barnes Meadow, B. Blue Lagoon, C. Ditchford, D. Draycote, E. Glebe Meadow, F. Pitsford, G. River Ise Meadows, H. Scrub Field, I. Twywell.

For the flower visitor species distribution between site type for 2007, there was no significant difference between the species distributions on the two types of site: (Figure 4.04) (Wilcoxon signed rank test; Restored landfill (*Median* = 1), Reference (*Median* = 1), $z = -0.24$, $r = -0.03$, $p = 0.81$). Of the species observed, there were 13 species present on restored landfill sites, but not reference sites; 20 species present on reference sites and not restored landfill sites; and 17 found on both. For the flower visitor species distribution for 2008, similarly there was no significant difference between the species distributions on the two types of site (Figure 4.05) (Wilcoxon signed rank test; Restored landfill (*Median* = 1), Reference (*Median* = 1), $z = -0.02$, $r < -0.01$, $p = 0.98$). For the analysis of species presence, there were 14 species present on restored landfill sites, but not reference sites, 13 species present on reference sites and not restored landfill sites, and 27 found on both.

The flower visitor species were assessed for abundance, frequency and presence on the types of site for 2007 (Figure 4.06). Thirteen of the 50 species were present on more restored landfill sites than reference sites; 20 of the 50 species were present on more reference sites than restored landfill sites; and 17 of the species were present on the same number of each type of site (Appendix 3). For 2008, Eighteen of the 53 species were present on more restored landfill sites than reference sites; 23 of the 53 species were present on more reference sites than restored landfill sites; and 12 of the species were present on the same number of each type of site (Figure 4.07) (Appendix 4).

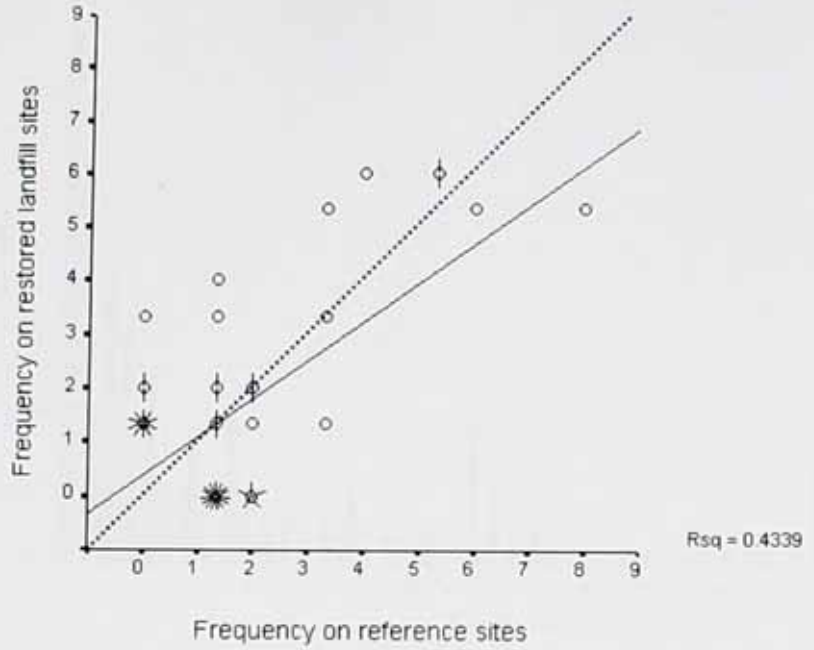


Figure 4.04 Frequency of flower visitor species present on restored landfill and reference sites 2007. Each point represents a species. Dashed line = 1:1, solid line = line of best fit. 13 species of flower visitor were above, 20 below and 17 on the line. 'Sunflowers'= multiple points, each petal represents an additional data point.

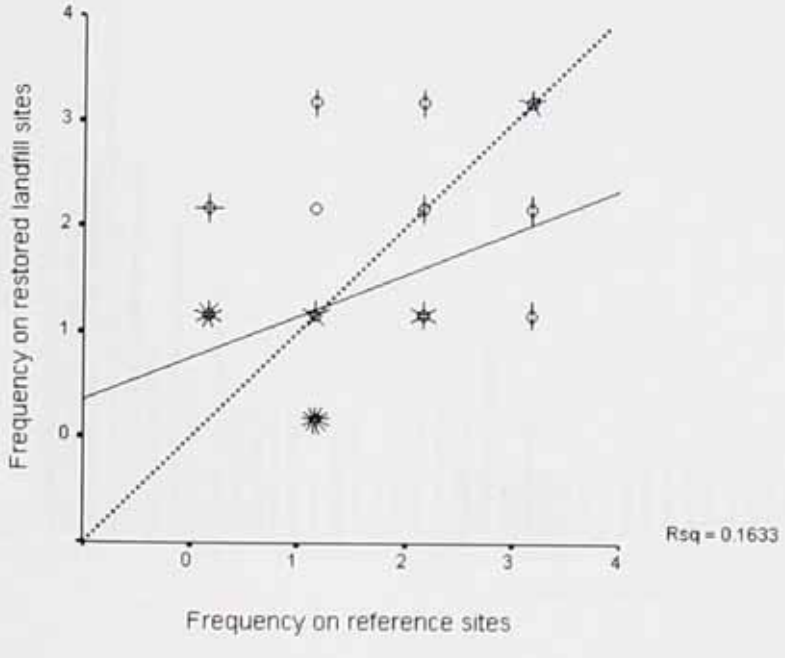
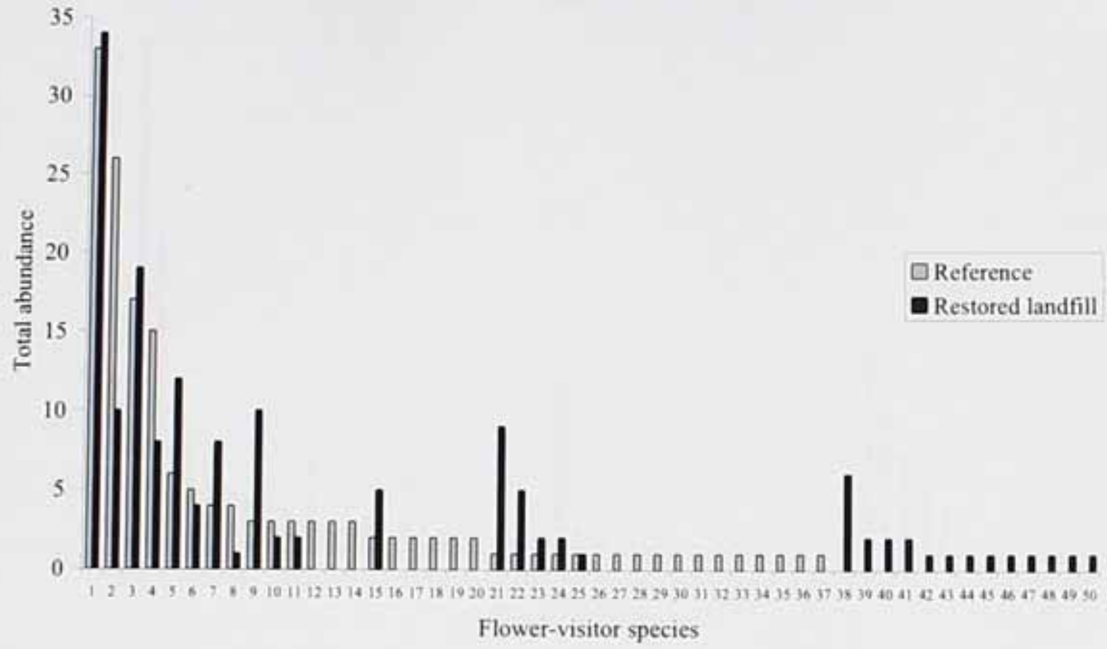


Figure 4.05 Frequency of flower visitor species present on restored landfill and reference sites 2008. Each point represents a species Dashed line = 1:1, solid line = line of best fit. 18 species of flower visitor were above, 23 below and 12 on the line. 'Sunflowers'= multiple points, each petal represents an additional data point.



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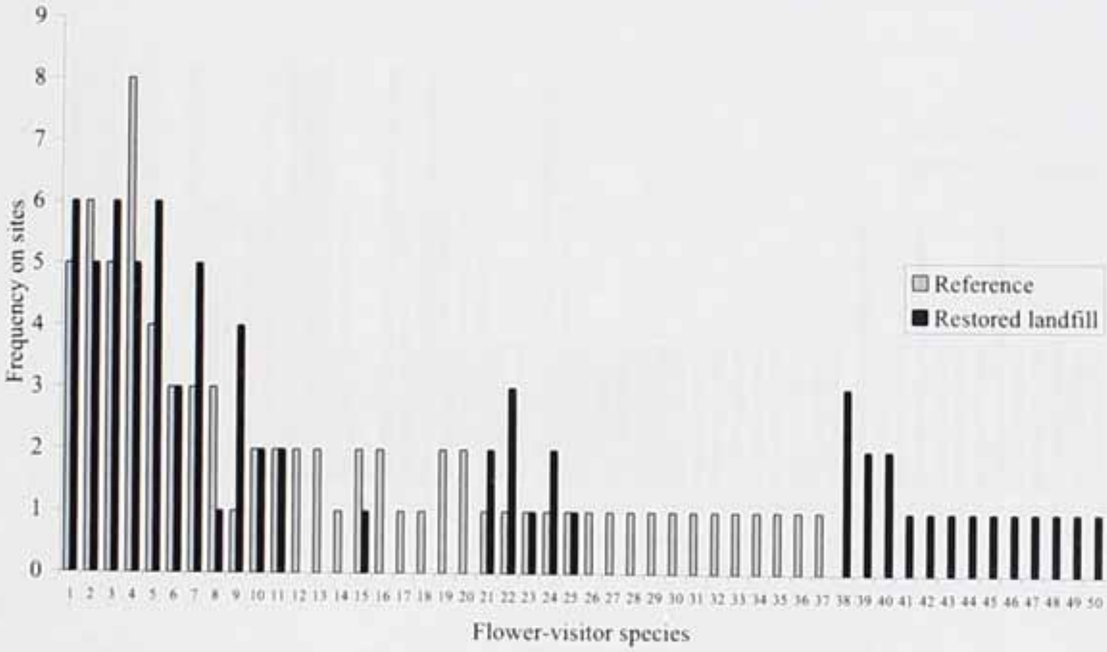
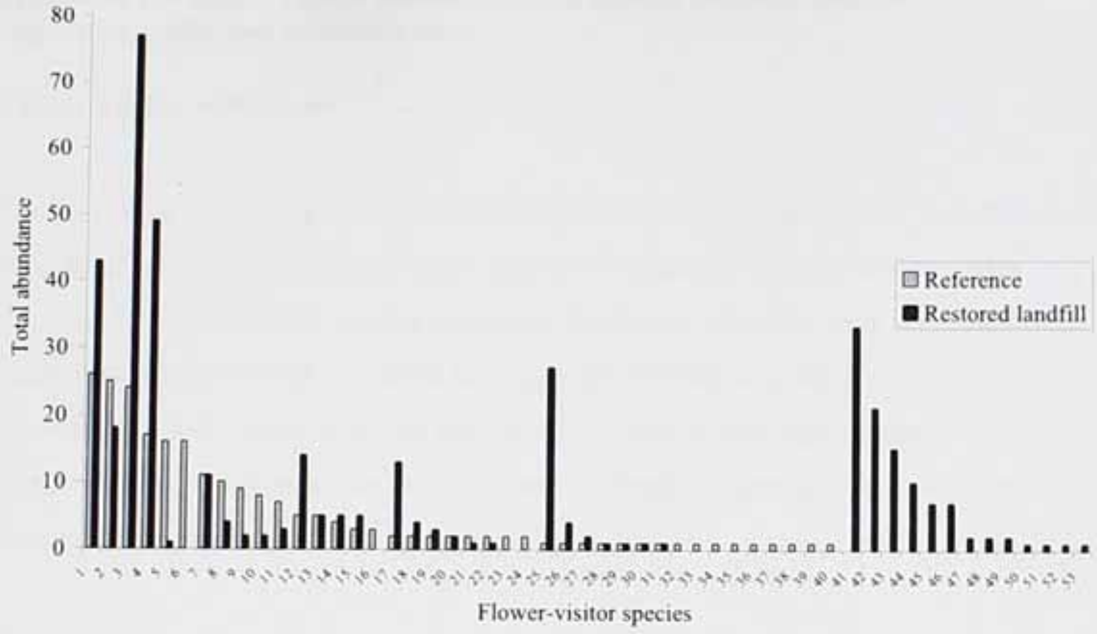


b)

Figure 4.06 Total flower visitor abundance and frequency on restored landfill and reference sites 2007 a) Total abundance b) Frequency on sites (Ranked according to reference site abundance). Species identities are listed in Appendix 3.



a)

b)

Figure 4.07 Total flower visitor abundance and frequency on restored landfill and reference sites 2008 a) Total abundance b) frequency on sites (Ranked according to reference site abundance). Species identities are listed in Appendix 4.

How does the flower visitor abundance and species richness compare between restored landfill and reference sites?

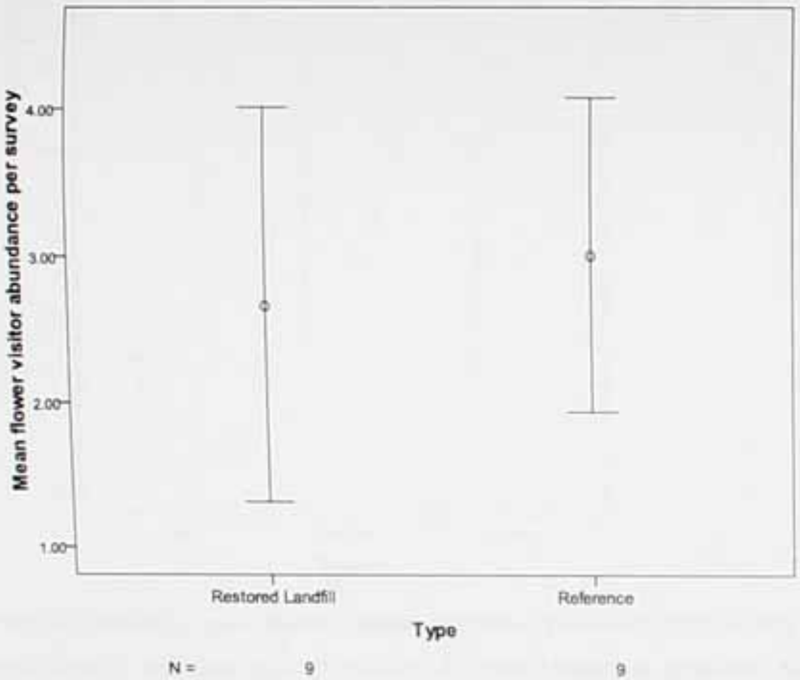
Flower visitor abundance

For the flower visitor abundance in 2007, there was no difference in the overall mean abundance of individuals recorded per survey on landfill sites and reference sites (Figure 4.08a). There was also no significant difference when the data were analysed seasonally (Figure 4.09a). For 2008 there was no difference in the overall mean abundance of individuals recorded per survey on landfill sites and reference sites (Figure 4.08b). There was also no significant difference when the data were analysed seasonally (Figure 4.09b).

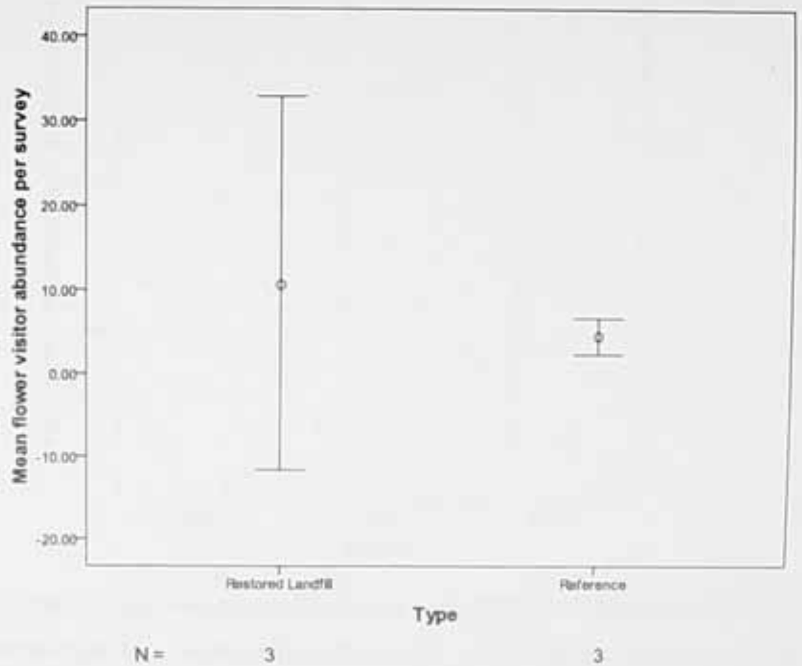
Flower visitor species richness

For both 2007 and 2008, there was no significant difference between the total flower-visiting insect species richness per site for restored landfill and reference sites (Figure 4.10), suggesting that both site types are supporting equally diverse flower-visiting insect faunas.

Flower visitor richness per survey on the landfill sites and reference sites was not significantly different for either 2007 or 2008 (Figure 4.11). The data were again assessed for seasonal variation and there was no significant difference between the site types within any season of both years (Figure 4.12), mirroring the abundance findings (above).

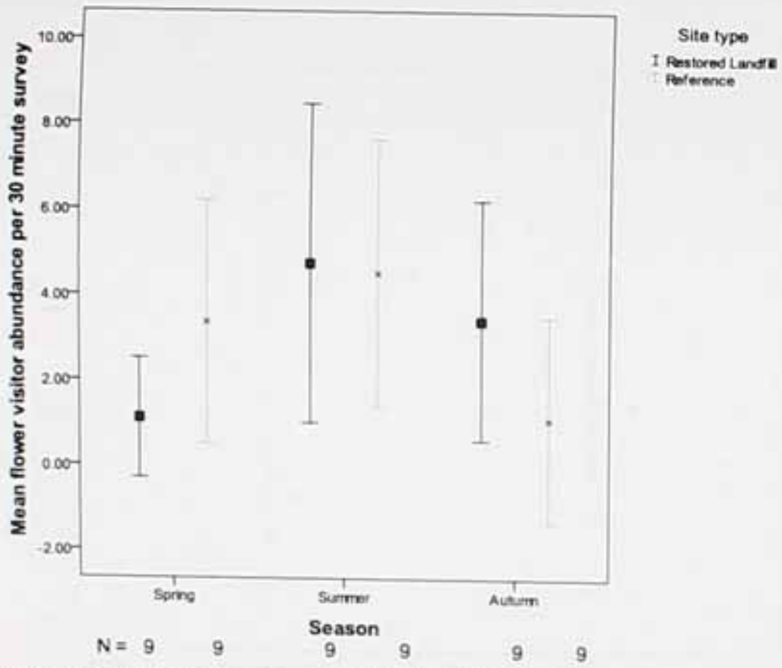


a)

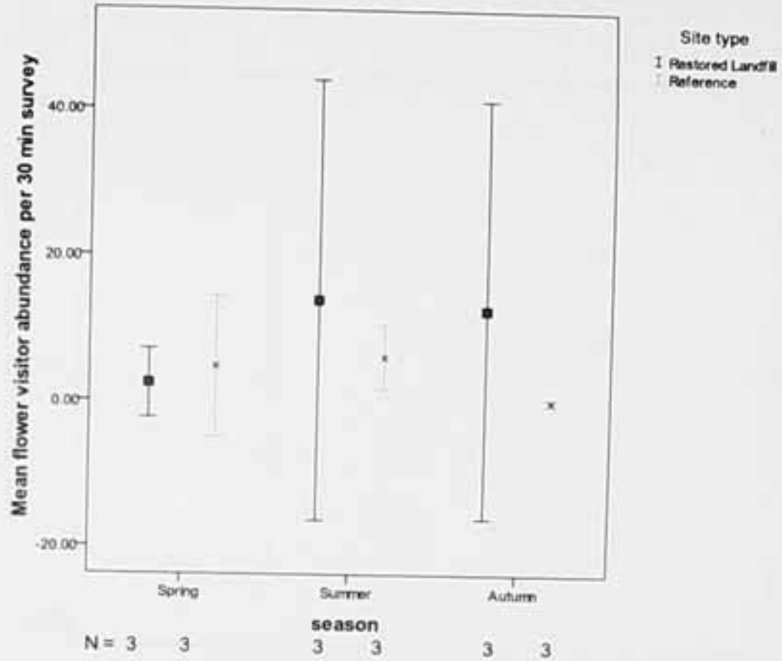


b)

Figure 4.08 Mean flower visitor abundance per survey for landfill sites and reference sites (± 95% Confidence Limits). N= sample sizes. a) 2007 Paired samples t-test (two-tailed) t=0.05, df=8, p=0.97 (survey = 30 minutes & 200m^2), b) 2008 Paired samples t-test (two-tailed) t=1.25, df=2, p=0.34 (Survey = 30 minutes & 600m^2).



a)



b)

Figure 4.09 Seasonal mean flower visitor individual abundance per survey per site for landfill sites and reference sites (± 95% Confidence Limits). N= sample sizes. a) 2007 (survey = 30 minutes & 200m^2) One-way ANOVA; Restored landfill sites across seasons $F_{2,17}$=2.81, p=0.09. Reference sites across seasons $F_{2,20}$=1.66, p=0.22. Paired samples t-test (two-tailed) Spring Landfill vs. Reference t= -1.31 df=5 p=0.25, Summer Landfill vs. Reference t=0.029 df=5 p=0.98, Autumn Landfill vs. Reference t= 0.38 df=2 p=0.74. b) 2008 (survey = 30 minutes & 600m^2) One-way ANOVA; Restored landfill sites across seasons $F_{2,8}$=1.25, p=0.35. Reference sites across seasons $F_{2,8}$=4.93, p=0.05. Paired samples t-test (two-tailed) Spring Landfill vs. Reference t= -1.73 df=2 p=0.23, Summer Landfill vs. Reference t=1.03 df=2 p=0.41, Autumn Landfill vs. Reference t= 1.85 df=2 p=0.21.

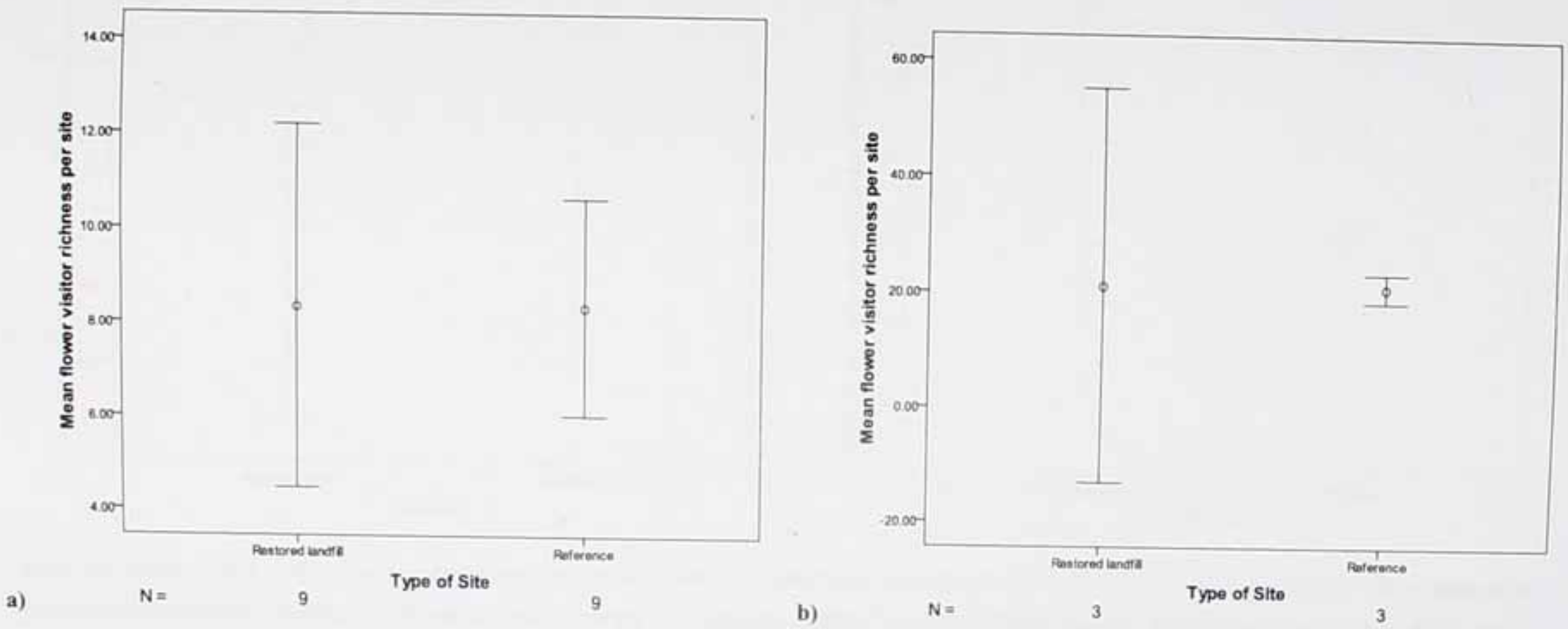
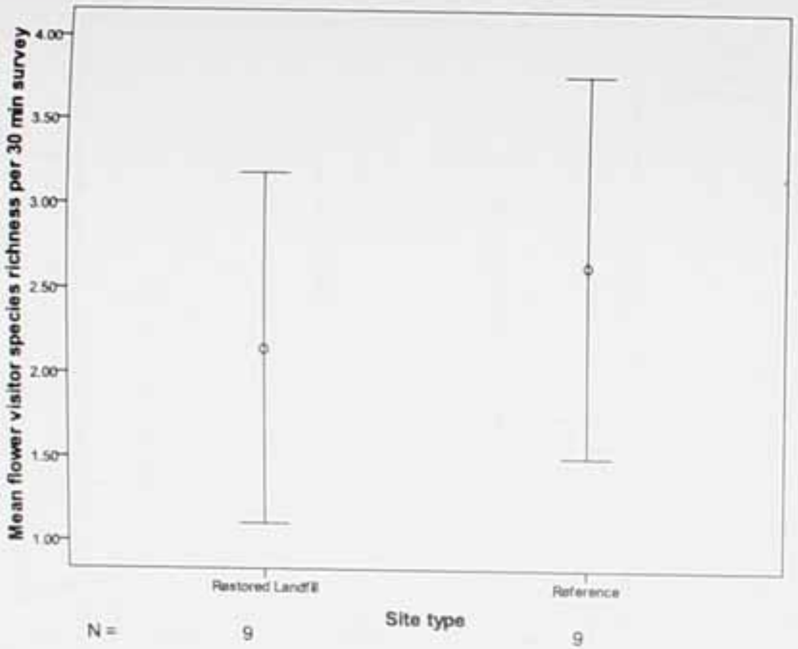


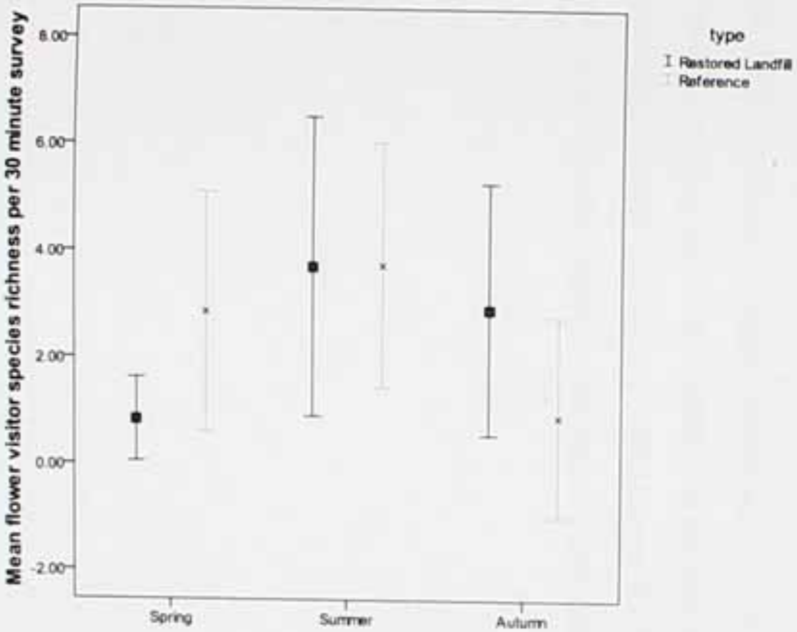
Figure 4.10 Mean annual total flower visitor species richness per site for restored landfill sites and reference sites (± 95% Confidence Limits). N=sample sizes. a) 2007 Paired samples t-test (two-tailed) t=0.00, df=8, p=1.00 (survey = 30 minutes & 200m^2), b) 2008 Paired samples t-test (two-tailed) t=0.00, df=2, p=1.00 (Survey = 30 minutes & 600m^2).



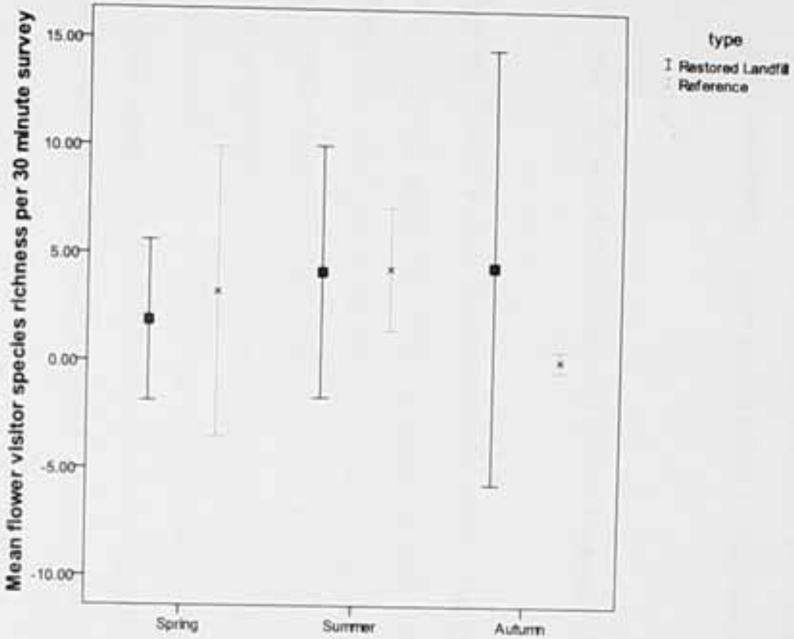
a)

b)

Figure 4.11 Mean flower visitor species richness per survey for restored landfill sites and reference sites (± 95% Confidence Limits). N=sample sizes. a) 2007 Paired samples t-test (two-tailed) t=-1.29, df=8, p=0.23 (survey = 30 minutes & 200m^2), b) 2008 Paired samples t-test (two-tailed) t=0.52, df=2, p=0.66 (Survey = 30 minutes & 600m^2).



a)



b)

Figure 4.12 Mean seasonal flower visitor species richness per survey for landfill sites and reference sites (± 95% Confidence Limits). N = sample sizes. a) 2007 (survey = 30 minutes & 200m^2) One-way ANOVA; Restored landfill sites across seasons $F_{2,17}$=3.12, p=0.07. Reference sites across seasons $F_{2,20}$=2.03, p=0.16. Paired samples t-test (two-tailed) Spring Landfill vs. Reference t= -1.60 df=5 p=0.17, Summer Landfill vs. Reference t=-0.24 df=5 p=0.82, Autumn Landfill vs. Reference t= 0.49 df=2 p=0.67. b) 2008 (survey = 30 minutes & 600m^2) One-way ANOVA; Restored landfill sites across seasons $F_{2,8}$=073, p=0.52. Reference sites across seasons $F_{2,8}$=5.00, p=0.05. Paired samples t-test (two-tailed) Spring Landfill vs. Reference t= -1.14 df=2 p=0.37, Summer Landfill vs. Reference t=-0.11 df=2 p=0.93, Autumn Landfill vs. Reference t= 1.76 df=2 p=0.22.

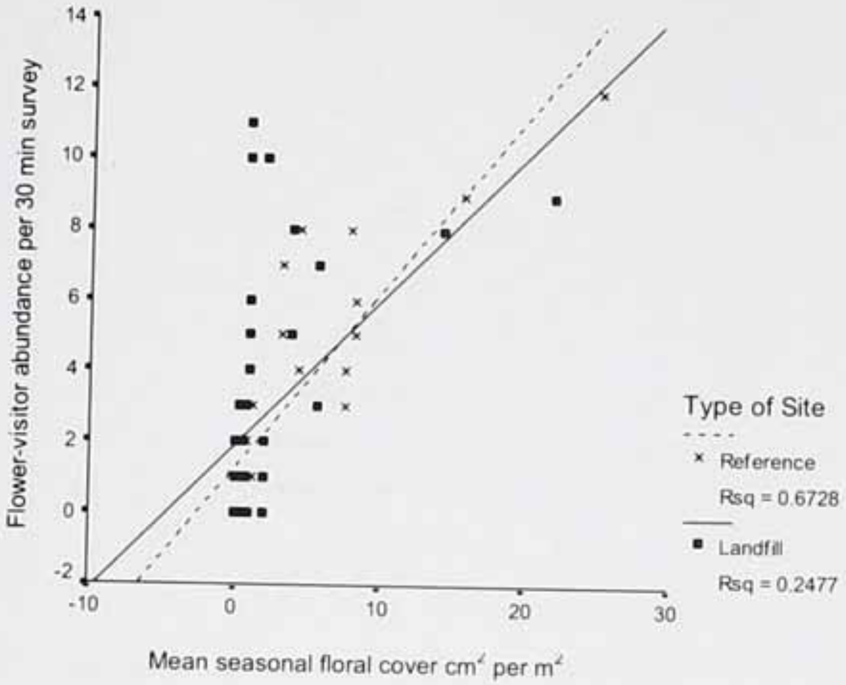
How does floral resource use compare between landfill and reference sites with regards to flower visitors and total on-site floral cover and richness of flowering plants?

Flower visitor abundance and mean seasonal floral cover

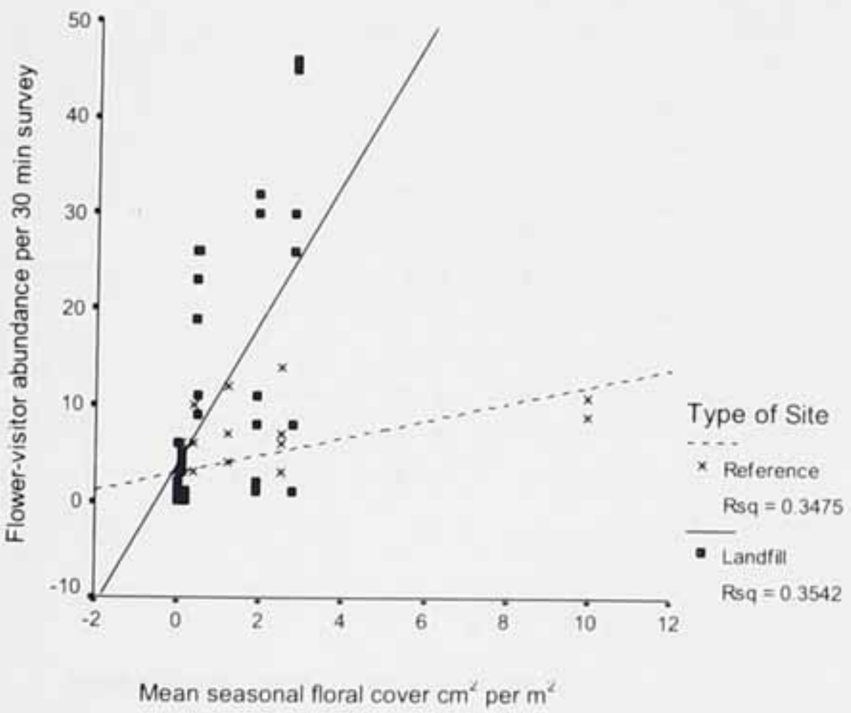
For the flower visitor abundance per survey and the mean seasonal floral cover, there was a significant positive relationship for both landfill and reference sites in 2007 and 2008. For 2007, the reference sites have a tighter correlation than do the restored reference sites (Figure 4.13a); for 2008 this was reversed (Figure 4.13b). The flower visitor abundance figures were higher for 2008 owing to the revised survey method being used. The same floral surveying method was used in both years, however, 2007 saw the sampling of more sites with lower sampling frequency, whilst in 2008 there were fewer sites sampled at higher sampling frequency.

Flower visitor species richness and richness of flowering plants

For the flower visitor species richness per survey and the mean seasonal richness of flowering plants, there was a significant positive relationship for both restored landfill and reference sites in 2007 and 2008 (Figure 4.14). For 2007 and 2008 the reference sites have tighter correlation than the restored landfill sites.

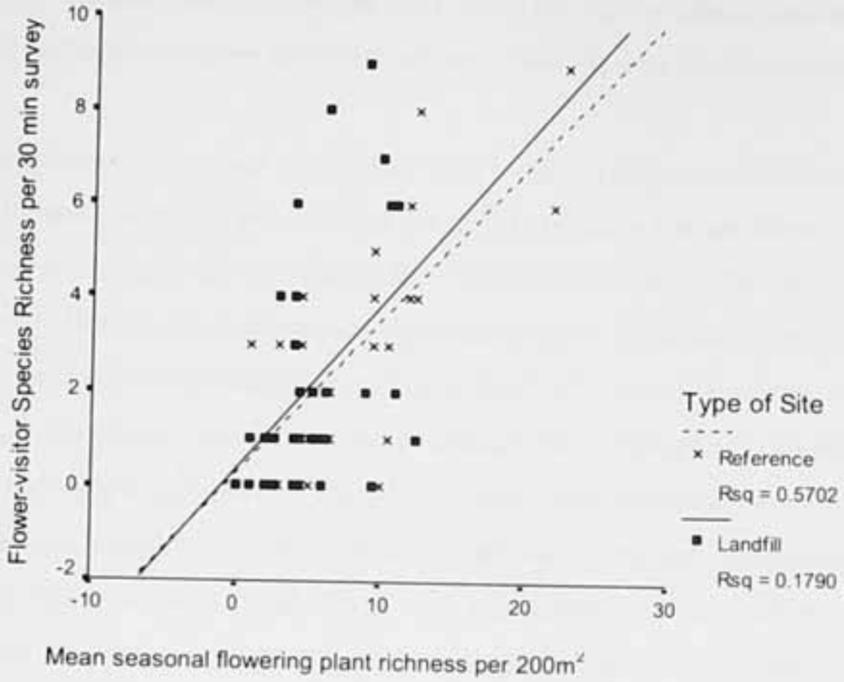


a)

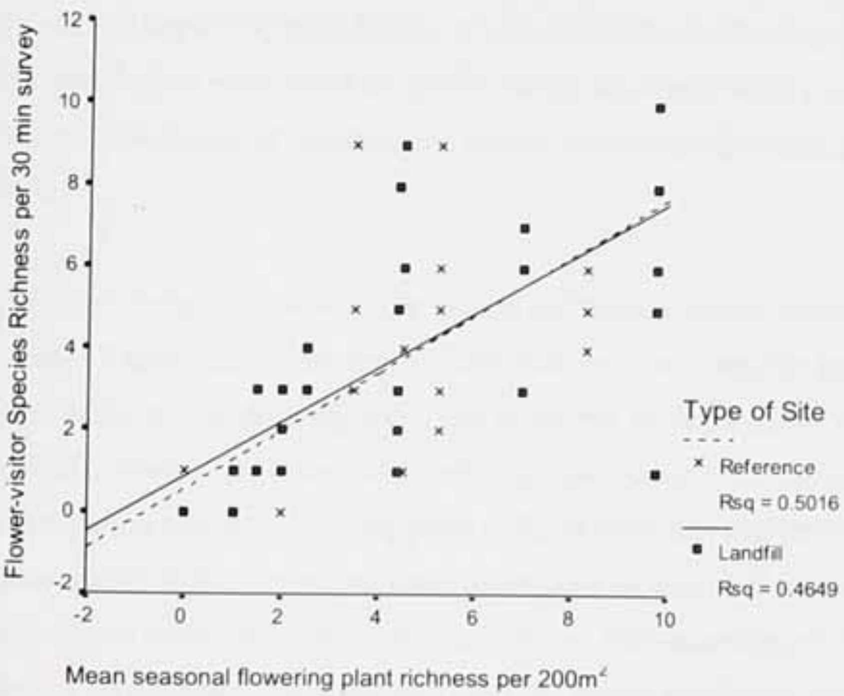


b)

Figure 4.13 Flower visitor abundance per survey correlated with mean seasonal floral cover (cm^2 per m^2). a) 2007 (Flower visitor survey = 30 minutes & $200m^2$). Landfill sites n = 44, Reference site n = 37. Spearman's rank correlation (Two-tailed) Landfill sites $r = 0.61$, $p < 0.001$; Reference sites $r = 0.83$, $p < 0.001$. b) 2008 (Flower visitor survey = 30 minutes & 600 m^2) Landfill sites n = 36, Reference site n = 30. Spearman's rank correlation (Two-tailed) Landfill sites $r = 0.70$, $p < 0.001$; Reference sites $r = 0.82$, $p < 0.001$.



a)



b)

Figure 4.14 Flower visitor species richness per survey correlated with on-site seasonal richness of flowering plants. a) 2007 (Flower visitor survey = 30 minutes & 200 m^2) Landfill sites n = 44, Reference site n = 38. Spearman's rank correlation (two-tailed) Landfill sites $r = 0.44$, $p < 0.01$; Reference sites $r = 0.72$, $p < 0.001$. b) 2008 (Flower visitor survey = 30 minutes & 600 m^2) Landfill sites n = 36, Reference site n = 30. Spearman's rank correlation (two-tailed) Landfill sites $r = 0.81$, $p < 0.001$; Reference sites $r = 0.82$, $p < 0 .001$.

How do habitat quality features determine the flower visitor assemblage? How do these features compare between restored landfill sites and reference nature sites?

Correlations were carried out for the flower visitor richness and abundance on restored landfill and reference sites, and habitat quality features and are shown in Table 4.04. The level of significant correlation has been adjusted using a Bonferroni correction to $p \leq 0.005$. With normal correlation significance ($p \leq 0.05$) in multiple tests, 1 in 20 would be expected to be significant, therefore with 11 x 7 tests, by random chance would see about 4 significant correlations due to chance alone. Therefore both *highly* significant ($p \leq 0.005$) and significant ($p \leq 0.05$) results have been described. For restored landfill sites the only highly significant positive correlation was between flowering plant and flower visitor richness (Table 4.05). Less significant positive correlations were found for both flower visitor richness and abundance, and availability of suitable bare earth and microstructures (Table 4.05). For the reference sites, the only highly significant correlation was negatively with the age of the sites (Table 4.05). Less significant positive correlations were found for flower visitor abundance with flowering plant richness and flower visitor richness; and negatively correlated with dead vegetation (Table 4.05).

For correlation between flower visitor groups and habitat quality variables on restored landfill sites, highly significant negative correlations were found for bees and the age of the sites, beetles and peak floral cover, and positively for bumblebees and bare earth (Table 4.06). There were other less significant positive correlations for habitat variables and insect groups namely; flowering plant richness with bees and hoverflies; max vegetation height with beetles and bumblebees; bare earth with hoverflies; features with bumblebees and hoverflies; and south facing slopes with bumblebees. For the correlations on reference sites there was only one highly significant positive correlation - between size of site and beetles (Table 4.07). There are other less significant negative correlations for habitat variables and insect groups namely; size of site with bees; bare earth with Other insects; and dead vegetation and south facing slopes with flies.

Table 4.05 Correlation of habitat quality variables with flower visitor species richness and abundance on restored landfill and reference sites for 2007 (n=9) (Pearson's correlation: two-tailed).

	Restored landfill sites				Reference sites			
	Flower visitor richness		Flower visitor abundance		Flower visitor richness		Flower visitor abundance	
Variable	**r**	**p**	**r**	**p**	**r**	**p**	**r**	**p**
Size of site	0.225	0.561	0.192	0.620	0.126	0.747	-0.002	0.996
Age	-0.407	0.278	-0.382	0.310	-0.207	0.594	-0.845	****0.004**
Flowering plant richness	0.852	****0.004**	0.724	***0.027**	0.044	0.910	0.705	***0.034**
Peak floral cover	0.128	0.743	0.311	0.415	0.051	0.896	0.587	0.097
Total plant richness	0.360	0.341	0.184	0.636	0.167	0.668	0.486	0.185
Max vegetation height	0.236	0.540	0.348	0.359	-0.171	0.660	-0.058	0.883
Bare earth / sandy soil	0.800	***0.010**	0.824	***0.006**	0.295	0.442	0.138	0.724
Features / holes	0.809	***0.008**	0.829	***0.006**	-0.191	0.622	0.647	0.060
Dead vegetation	0.646	0.060	0.540	0.134	-0.714	***0.031**	0.066	0.866
South facing slopes	0.485	0.185	0.621	0.074	-0.266	0.488	0.140	0.719
Hedges / shrubs	-0.116	0.765	-0.084	0.830	0.011	0.977	0.433	0.245

* Significant at $p \leq 0.05$

** Significant at $p \leq 0.005$, due to Bonferroni correction.

Table 4.06 Correlation of habitat quality variables with species richness of flower visitor groups on restored landfill sites for 2007 [1] (n=9) (Pearson's correlation: two-tailed).

Variable	Bees		Beetles		Bumblebees		Flies		Hoverflies		Other	
	r	p	r	p	r	p	r	p	r	p	r	p
Size of site	0.375	0.321	-0.216	0.577	-0.081	0.836	0.389	0.300	0.160	0.681	0.140	0.719
Age	-0.871	****0.002**	0.146	0.708	-0.330	0.385	0.007	0.985	-0.427	0.252	0.035	0.929
Flowering plant richness	0.741	***0.022**	-0.099	0.799	0.501	0.170	0.445	0.230	0.788	***0.012**	0.390	0.299
Peak floral cover	-0.374	0.322	0.915	****0.001**	0.623	0.073	0.223	0.565	-0.182	0.640	-0.216	0.577
Total plant richness	0.250	0.517	0.285	0.458	0.434	0.243	0.622	0.074	0.151	0.698	-0.202	0.602
Max vegetation height	-0.496	0.175	0.813	***0.008**	0.704	***0.034**	-0.133	0.733	0.027	0.945	0.117	0.765
Bare earth / sandy soil	0.503	0.168	0.341	0.370	0.851	****0.004**	0.093	0.811	0.701	***0.035**	0.263	0.494
Features / holes	0.262	0.496	0.357	0.345	0.750	***0.020**	-0.025	0.948	0.681	***0.043**	0.627	0.070
Dead vegetation	-0.228	0.555	0.249	0.519	0.644	0.061	0.022	0.955	0.617	0.077	0.546	0.128
South facing slopes	0.049	0.900	0.609	0.082	0.745	***0.021**	-0.043	0.912	0.346	0.362	0.131	0.737
Hedges / shrubs	-0.611	0.080	0.575	0.105	0.350	0.356	-0.055	0.888	-0.226	0.558	-0.222	0.566

* Significant at $p \leq 0.05$

** Significant at $p \leq 0.005$, due to Bonferroni correction.

1. Butterflies not correlated as none found.

Table 4.07 Correlation of habitat quality variables with species richness of flower visitor groups on reference sites for 2007 [1] (n=9) (Pearson's correlation: two-tailed).

Variable	Bees		Beetles		Bumblebees		Butterflies		Flies		Hoverflies		Other	
	r	p	r	p	r	p	r	p	r	p	r	p	r	p
Size of site	-0.671	***0.048**	0.841	****0.005**	0.083	0.831	-0.058	0.882	0.311	0.415	0.261	0.497	0.328	0.389
Age	-0.084	0.830	-0.279	0.467	-0.321	0.400	0.242	0.530	-0.055	0.889	-0.267	0.488	-0.349	0.357
Flower rich.	0.156	0.688	-0.214	0.581	0.188	0.629	0.083	0.832	-0.181	0.642	0.226	0.559	-0.358	0.344
Pk. floral cover	0.432	0.245	-0.304	0.427	0.159	0.683	0.004	0.991	-0.027	0.944	0.019	0.962	-0.539	0.134
Total plant rich.	0.061	0.877	-0.044	0.910	0.291	0.447	0.323	0.397	-0.026	0.946	0.238	0.538	-0.588	0.096
Max veg. height	-0.168	0.666	-0.202	0.603	-0.424	0.255	0.015	0.970	0.287	0.455	-0.029	0.941	-0.584	0.099
Bare/sandy soil	0.303	0.428	-0.354	0.351	0.500	0.170	0.539	0.134	-0.316	0.407	0.508	0.163	-0.707	***0.033**
Features / holes	0.197	0.612	-0.229	0.553	-0.081	0.836	-0.332	0.382	-0.103	0.793	0.082	0.833	-0.459	0.214
Dead veg.	-0.035	0.929	-0.510	0.160	-0.433	0.244	-0.436	0.241	-0.822	***0.007**	-0.257	0.505	0.000	1.000
South facing slope	-0.177	0.649	-0.026	0.947	0.182	0.639	0.016	0.968	-0.807	***0.009**	0.093	0.813	0.103	0.792
Hedges / shrubs	0.613	0.079	-0.169	0.665	0.191	0.623	-0.267	0.487	-0.030	0.939	-0.266	0.488	-0.135	0.729

* Significant at $p \leq 0.05$

** Significant at $p \leq 0.005$, due to Bonferroni correction.

Discussion

Fifty species of flower visitor were recorded in 2007 and 53 in 2008, all of which were common species (See Appendix 4). Comparison with previous studies is difficult given the different sites and different methods employed but it is a useful indication of the relative species richness of flower-visiting insects on the restored and reference sites (Table 4.08). The results for 2007 and 2008 are relatively low compared to the other studies shown. However, given the spread of the results they can be seen as similar to other studies from England (Figure 4.10 & Table 4.08). With relevance to the conservation potential of the restored landfill sites it should also be noted that the species richness significantly increased when the species present across all sites was determined; for 2007: restored landfill sites = 30, reference sites = 37; for 2008: restored landfill sites = 41, reference sites = 40. It should however be re-iterated that this is predominantly a comparative study between the restored landfill and the reference sites rather than an attempt to assess the total number of species. The important issue is that equal effort of sampling has been used on all sites.

Table 4.08 Examples of species richness for community surveys of insect flower visitors.

System	Location	Insects	Plants	Published
Mediterranean shrubland.	SW Spain	179.0	26.0	(Herrera, 1985)[#]
Grassland	Cass, New Zealand	139.0	41.0	(Primack, 1983)
Waste ground	Denmark	82.0	26.0	(Olensen, *unpub.*)[+]
Restored hay meadows	SW England	40.5*	27.0*	(Forup and Memmott, 2005)
Restored Heathland	Devon, England	36.5*	6.8*	(Forup et al., 2008)
Old Heathland	Devon, England	35.6*	5.3*	(Forup et al., 2008)
Old hay meadows	SW England	30.5*	21.0*	(Forup and Memmott, 2005)
Restored landfill sites 2008	East Midlands UK	21.0*	18.0*	**(This thesis)**
Reference sites 2008	East Midlands UK	21.0*	17.3*	**(This thesis)**
Restored landfill sites 2007	East Midlands UK	8.3*	16.8*	**(This thesis)**
Reference sites 2007	East Midlands UK	8.3*	19.1*	**(This thesis)**

*** - Mean per site given.** [#] cited in Jordano (1987); [+] cited in Olesen and Jordano (2002)

Examining the distribution of groups and total number of species between the restored landfill and reference sites (Table 4.02), it is seen that in 2007 there were more species supported on the reference sites than the restored landfill sites (Figure 4.04). In 2008 however, following more intensive sampling on fewer sites, the restored landfill sites were found to support slightly more (Figure 4.05). Within the groups there were varying numbers of species between the two types of sites (Table 4.02 and Figure 4.02). For example, in 2007 there were six species of butterflies found on the reference sites, but only one on the restored landfill sites. In 2008, there were six species of bees on the restored landfill sites but only two on the reference sites. In both years hoverflies were present in greater abundance and richness on the restored landfill sites. Hoverflies have been found to be positively correlated with resource heterogeneity such as species richness of flowering plants (Meyer et al., 2009), and the earlier successional state of the restored landfill sites may provide this (see Chapter 3).

Predictions for the similarity of species richness of different groups on the restored landfill and reference sites relate to their dispersal ability and habitat specific requirements. Those groups with greater dispersal abilities and less specific habitat requirements would show an increased similarity between the two sites. This has been shown for the bumblebee group, with little difference between the species richness on the two types of site (Table 4.02). Bumblebees are not generally habitat specialists (Goulson, 2003a). Bumblebee foraging ranges and hence dispersal ability are at least several hundred metres and even several kilometres from their nests (Dramstad, 1996; Osborne et al., 1999; Goulson and Stout, 2001; Kreyer et al., 2004; Knight et al., 2005; Westphal et al., 2006; Osborne et al., 2008); so it would be expected they disperse easily across the landscape and colonise or utilise new habitat sites readily. Those groups with poorer dispersal abilities and more specific habitat requirements would be expected to see a greater difference between the two types of site. This idiosyncratic result between different groups has been seen for butterflies. This difference in species richness between sites may be attributed to site-specific requirements of particular species, butterflies often requiring larval-specific host plants (See Table 4.03) and their dispersal distances are relatively poor, 100-200m (Krauss et al., 2003; Cant et al., 2005). There were also differences for the species richness of the fly group (Diptera, non-Syrphidae), with more species present on the reference than the landfill sites. However, there is little information available regarding their dispersal abilities and habitat requirements, which highlights that further research is required for this over-looked group of flower-visiting insects.

There clearly may be specific habitat requirements for some species, for example oligolectic bees, but further work is required to investigate this in detail. This has not been attempted within this study in detail owing to limited time and resources available. Overall, the species richness of flower-visiting insect assemblage was no different for the restored landfill or reference sites (Figures 4.10 & 4.11).

With regards to species distributions between the site types, there was approximately one quarter of species unique to restored landfill sites and reference sites and one half of species present on both (Figures 4.06 & 4.07). This is similar to the situation for plants (Chapter 3). The ordination analysis shows a high level of overlap between the types of sites, with a few outliers, as described previously (Figure 4.03). However, when the data were assessed for the frequency of occurrence on sites, species were more likely to be present on a greater number of reference sites than landfill site (Figures 4.06 & 4.05). The restored landfill sites are newly created habitats and so species have to disperse to the sites from the surrounding landscape or from nearby areas of semi-natural vegetation.

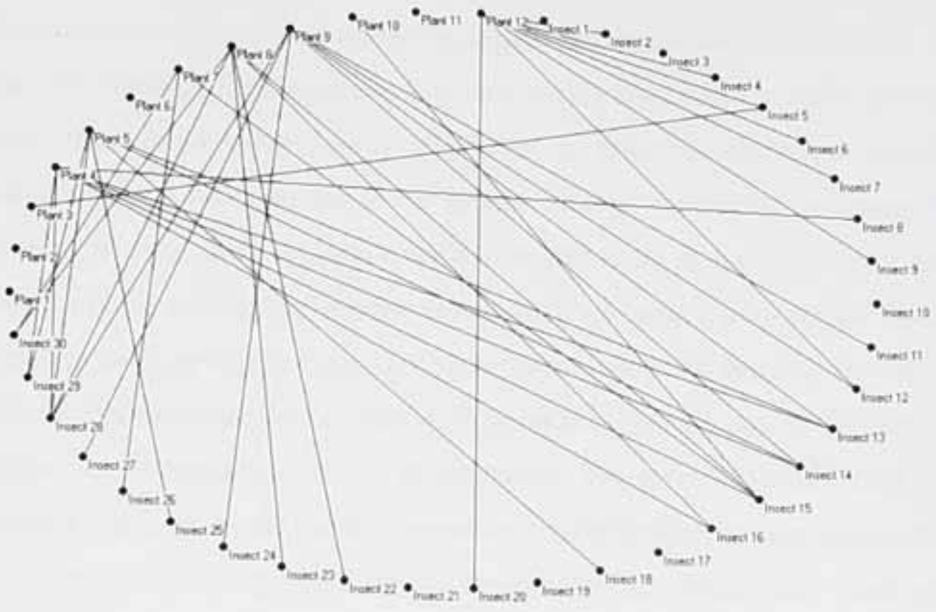
The expectation was that the restored landfill sites would have lower flower visitor abundance and species richness than the reference sites; however there was no difference for annual means (Figures 4.08 & 4.10). It appears that the restored landfill sites were supporting a comparable flower-visiting insect assemblage. Possible reasons for this may be that in early succession, as found on the restored landfill sites, there is generally a greater richness of plant pioneer species (Denslow, 1980) and consequently a greater richness and abundance of flower visitors (Potts et al., 2003b). However, succession theory predicts that highest species richness in mid-successional stages of "intermediate disturbance" (Connell, 1978; Brown and Southwood, 1987), therefore there may be no difference between the two types of site.

The findings for this study are similar to previous research comparing restored and reference sites (Table 4.08). On restored versus established hay meadows, for example, few flower-visiting insect species were shared among the sites, but the species richness and abundance were the same between the old and new sites (Forup and Memmott, 2005). On four pairs of restored and ancient heathlands, the restored sites had fewer insect species than their pairs in 2001, but more species in 2004 (Forup et al., 2008). However, it should be noted that in both of these studies the restored sites were directly targeted at recreating the same vegetation structure as that of the reference sites, hay meadows and heathlands respectively. This is clearly not the case in the present study.

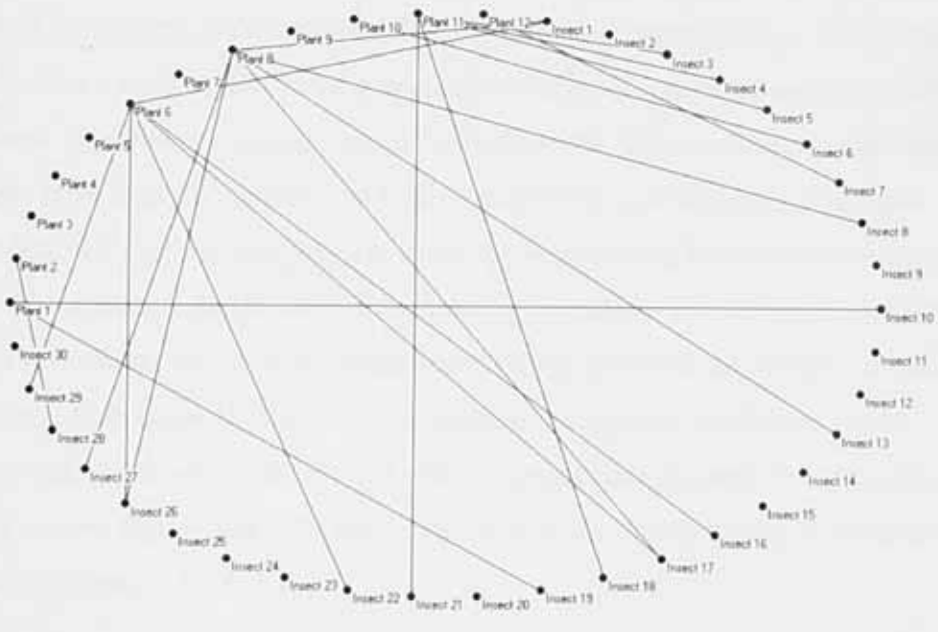
Does this show that the restored landfill sites are as good as the reference nature sites at supporting pollinating flower-visiting insects? Yes, the annual and peak summer activity was equally matched (Figures 4.09, 4.10, & 4.12). They had the same mean species richness per site, peak seasonal richness and abundance. It appears unlikely that the restored landfill sites are acting as sinks for flower-visiting insect populations, although without evidence of nesting on-site, it is unclear as to how the assemblage is utilising them. Those groups of flower-visiting insects supported well, in terms of individual abundance and species richness, on restored landfill sites are similar to those on the reference sites and includes bumblebees, hoverflies and flies (Tables 4.02 & 4.04 and Figure 4.02). These groups may be generally more abundant throughout the local agricultural landscape. The most abundant species of flower visitor, present on the majority of sites were common, abundant species such as *Apis mellifera*, *Bombus terrestris/lucorum*, *Bombus pascuorum*, *Bombus lapidarius* and *Calliopum* spp. (Table 4.04) (Goulson et al., 2005).

This presence of common and abundant species shared between types of sites may be indicative of the "homogenisation" of pollinating flower visitor insects found in England. Those species of insect which are common and able to adapt to the intensive agricultural landscape may be increasingly more common throughout the rural landscape. Conversely, those species which are rare and doing poorly within the landscape may be becoming rarer or extinct. Reasons for this would need further research to determine, but the general parallel homogenisation of the landscape, with national agricultural practices and grown crops, is likely to play a significant role. This phenomenon has been recorded recently for woodland plant species (Keith et al., 2009). They found that woodland sites were losing their individual characteristics, and that although the number of species has remained the same, there was a significant loss in diversity. This biotic homogenisation clearly has large implications for the conservation of biodiversity. It can easily be missed when relatively small scale studies are looked at in isolation and more work is required to examine species presence and abundance on habitats across the landscape and nationally.

Populations of flower-visiting insects may be using the restored landfill sites as 'stepping stones' between more suitable habitats, or even as meta-habitats themselves. The connectivity between habitat sites has the possibility of mitigating the problems of fragmentation and size of habitats, by enabling the flow of individuals through the landscape (Beier and Noss, 1998; Falcy and Estades, 2007). Supporting population reserves of flower-visiting pollinating insects has anthropogenic advantages in the services they provide for agricultural and native plants. In a study assessing the design of agricultural landscapes for pollination service provision, small habitats interspersed throughout the landscape have been found to be optimal (Brosi et al., 2008), and the use of native habitats within the landscape has been found beneficial for the conservation of flower-visiting insects (e.g. Franzén and Nilsson, 2008; Ricketts et al., 2008). It has been argued previously that large areas of suitable habitat are required for species conservation, however, if flower-visitor species are able to readily travel between patches then this may not be necessarily the case (Goulson, 2003a). The importance of having numerous sites has been shown within this study, the restored sites supported a mean of 8-20 species per site, but across them all 30-40 species in total (Table 4.08). Research has indicated that native flower visitors may provide all of the pollination services required if there is 40% of the landscape as natural habitat; but even a more realistic 10% can provide a significant proportion of pollination services (Kremen et al., 2004). Across the Midlands and the UK there are a significant number of landfill sites which could be restored to meet this target (Chapter 6 - Figure 6.01).



a)



b)

Figure 5.02 Flower-insect visitor interaction structure in 2008 a) Wootton restored landfill site, b) Barnes Meadow reference site (Same plant and insect species identities were used for both interaction structures, where no link is present this refers to a species found on the comparison-sites, refer to Appendix 8 for species lists).

(Table 4.06). There were other less significant positive correlations for habitat variables and insect groups namely; flowering plant richness with bees and hoverflies; max vegetation height with beetles and bumblebees; bare earth with hoverflies; features with bumblebees and hoverflies; and south facing slopes with bumblebees. For the correlations on reference sites there was only one highly significant positive correlation between size of site and beetles (Table 4.07). There are other less significant negative correlations for habitat variables and insect groups namely; size of site with bees; bare earth with other insects; dead vegetation with flies; and south facing slopes with flies. Reasons for the observed relationship for bees may relate to their preferential floral resource species being present on younger sites. The relationship between bumblebees and the presence of bare earth is not clear as they do not typically nest in the earth, but may use empty small mammal holes, or in vegetation. The restored landfill sites do undergo some subsidence as the waste compresses and degrades, so opening cracks suitable for bumblebee nesting in the surface topsoil. The only significant relationship for the reference sites was negatively for beetles and the size of sites.

The causal reasons for the significant correlations have not been assessed here and further work is required to determine them; with a new research design, giving more sites and/or less variables. Different species are clearly idiosyncratic with their habitat requirements. For flower-visiting insects, their dispersal abilities clearly mean that habitat sites have fuzzy boundaries and they are not restricted to only using those resources from on-site. Instead they are using the surrounding habitat-stew, from which to supply their resource needs, be it, food, nesting or mating. The restored landfill sites are indeed themselves part of this habitat-stew. Given that both the restored landfill and reference sites both show similar levels of correlation between the flower visitor abundance with floral cover, and flower visitor species richness and flowering plant richness; it can be argued that they are being used by the flower visitor population in a similar way (Figures 4.13 & 4.14).

In conclusion, the restored landfill sites are supporting as rich and abundant flower visitor fauna as the reference sites. The same groups of insects are found on both types of sites, and there is significant overlap in species and abundance between the two types of site. Some groups of insects, namely hoverflies, are more species rich and abundant on the restored landfill sites than on the references sites, whilst others, namely butterflies, are less species rich and abundant than on the reference sites. The most abundant species are common ones and found on both types of sites. The restored landfill site restoration can clearly be seen as successful regarding the flower-visiting insects supported, given that the majority of reference sites were managed nature reserves. These sites have the potential to play a significant role in the conservation effort for the common flower visiting insects now in decline.

Results summary

In conclusion, the results of this part of the study can be summarised as follows:-

- No difference was found between the restored landfill and reference sites mean flower-visiting insects' species richness or abundance.
- No difference was found between the restored landfill and reference sites flower-visiting insects' species frequency on sites.
- There was no significant difference found between the flower visitor abundance or species richness per site for restored landfill and reference sites in any season.
- Hoverflies were the most species rich group of flower-visiting insects found on the restored landfill sites, and also the richest group on the reference sites.
- Hoverflies, bumblebees and flies were the groups with the greatest flower visitor abundance on the restored landfill sites, and also the most abundant groups on the reference sites.
- The most abundant species of flower visitor found on the majority of reference sites in both years were: *Bombus terrestris / lucorum*, *Bombus pascuorum*, *Bombus lapidarius* and *Calliopum* spp.
- Restored landfill and reference sites had similar relationships between their flower-visiting insect's richness and abundance and floral abundance and richness.
- The only habitat quality variable which flower visitor richness significantly correlated with was flowering plant richness.

Figure 4.15 ***Bombus lucorum*** **feeding on a Spear thistle on Harlestone restored landfill site.**

Chapter

5

The flower-insect interaction web on restored landfill sites

The flower-insect interaction web on restored landfill sites

"The real voyage of discovery consists not in seeking new landscapes but in having new eyes."
Marcel Proust (1871 - 1922)

"I have yet to see any problem, however complicated, which, when you looked at it the right way, did not become still more complicated."
Paul Alderson (b.1926)

Summary

This chapter describes the plant-flower-visitor interaction structure found on restored landfill sites in relation to their success in supporting assemblages of plants and flower visitors. The plant-flower-insect interactions were compared on nine paired sites of restored landfill sites and reference nature sites using standard belt transect methods. The bipartite interaction webs were analysed for uniqueness of interactions, core species, connectance and nestedness.

The restored landfill and reference site pairs shared few species within their interaction matrices and fewer actual species-species interactions. There were also few core species shared between the sites. There was no difference between connectance on the restored landfill sites and the reference sites, though the connectance levels were generally higher than most published network studies for assemblages of plants and pollinators. There was no difference in the generalisation levels of plants and insects between site types and the plants were more generalised than the flower-visiting insects. Nestedness within the restored landfill sites was similar to that of the reference sites, but generally lower than published data sets.

The flower visitor interactions were analysed by their network attributes in order to verify structural and functional variation between restored landfill sites and reference

sites of similar habitat conditions. The advantage of this functional approach in the evaluation of community structural attributes within restoration outcomes is that although the species differ, the underlying ecological process occurring on-site is apparent.

Introduction

Terrestrial restoration projects typically focus on the dominant plant species of the target community in the hope that natural processes will subsequently develop and move the community towards complete restoration (Palmer et al., 1997). It is difficult to define an exact target community for the restoration process, owing to ecologists rarely possessing complete biological records of the structure of an ecological community or even the region's complete species diversity. There has therefore been a move towards recognizing that most ecosystems are dynamic and hence restoration goals and assessments cannot be based on static attributes (Hobbs and Harris, 2001). Moreover, it is even harder to evaluate afterwards whether the restoration has been successful. In planning and evaluating restoration projects a purely structural focus is inadequate, as it does not show ecosystem functioning on the habitat: an alternative way which does, is to consider species interactions (Ehrenfeld and Toth, 1997).

The study of biological interactions in relation to restoration of habitats has concentrated on processes such as herbivory (e.g. Opperman and Merenlender, 2000; Sweeney et al., 2002; Ruhren and Handel, 2003), predation (e.g. Grimm and Backx, 1990; Olsson et al., 2002; Hartung and Brawn, 2005) and seed dispersal (e.g. Orth et al., 1994; Wunderle, 1997; Donath et al., 2003). Studies of the mutualistic interactions of assemblages of plants and pollinators include restored hay meadows (Forup and Memmott, 2005b), grassland (Maccherini et al., 2009), fragmented dry forests (Aizen and Feinsinger, 1994), prairies (Reed, 1995) and heathlands (Forup et al., 2008). Plant-pollinator interactions can play an important role in habitat restoration (Handel et al., 1994; Montalvo et al., 1997; Forup and Memmott, 2005b; Cortina et al., 2006; Falk et al., 2006). Once a vegetation community is restored then it is often assumed that species interactions such as pollination will be automatically reinstated. The use of species interaction webs allows the pollination systems of pristine and restored habitats to be compared, avoiding obvious discrepancies such as differing plant communities (Memmott, 1999).

Restoration of damaged ecosystems or the creation of new ones usually emphasise, and are measured in relation to, structural aspects of biodiversity such as species richness and abundance (Ruiz-Jaen and Mitchell Aide, 2005). Problems with using this approach are that there are often no records as to the target species; or different species are present on the reference site being used. An alternative approach is to emphasize functional aspects such as interactions between species, and ecological networks provide a powerful way of assessing the outcome of restoration programmes (Forup et al., 2008). The recovery of biological interactions in restored habitats is critical for their long-term functioning (Ruiz-Jaen and Mitchell Aide, 2005), processes such as nutrient cycling or herbivory are important to assess as they indicate the functionality of a habitat. However, ecological processes are not measured as frequently as diversity and vegetation structure (Ruiz-Jaen and Mitchell Aide, 2005). Comparing the pollination linkages found on the restored sites and reference sites can test whether ecological restoration has been successful (Montalvo et al., 1997; Palmer et al., 1997; Neal, 1998). The assessment of interaction structure within a plant-flower visitor assemblage allows a 'snapshot' of the functioning of an ecosystem, and this research has recently come to the fore (Forup and Memmott, 2005b; Fontaine et al., 2006a; Nielsen and Bascompte, 2007; Santamaria and Rodriguez-Girones, 2007; Forup et al., 2008; Dupont et al., 2009; Kaiser-Bunbury et al., 2009). When interaction webs are analysed in terms of connectedness and nestedness they may show the relative robustness of the system (Dunne et al., 2002). Assemblages of plants and pollinators are complex and dynamic (Waser et al., 1996), and assessing them in this functional approach is a useful measure in examining the wider ecological system.

Mutualistic associations between species in a community can be viewed as interaction networks, webs and matrices, all of which show that these types of relationships are highly plastic in space and time (Fortuna and Bascompte, 2006; Bascompte and Jordano, 2007; Alarcón et al., 2008; Fontaine et al., 2008; Petanidou et al., 2008), and may be robust to disturbance (Dunne et al., 2002; Potts et al., 2003b; Fontaine et al., 2006b; Vulliamy et al., 2006). For example, in multi-year studies of assemblages of plants and pollinators in Mediterranean habitats in Greece (Petanidou et al., 2008) and California (Alarcón et al., 2008), there were differences in both species composition and patterns of interaction from year to year, but network properties such as connectedness and nestedness remained constant. Patterns in pollination interactions can be analysed with a linked food web approach or in a nested complex system approach (Lewinsohn et al., 2006; Forup et al., 2008). Studies involving total communities of interacting plants and flower visitors are much rarer than those looking at single plants or involving guilds of plants or pollinators (Waser et al., 1996; Olesen and Jordano, 2002). An observation which has emerged from viewing interactions as nested assemblages of species is that there is often a central core of generalist species that interact with each other, and in so doing support more specialised species (Bascompte et al., 2003; Dupont et al., 2003; Ollerton et al., 2003; Olesen et al., 2007). If pollination biologists really want to understand how communities are structured and functioning, then all flower visitors must be included. Examining the entire pollination interaction web within a community avoids potential bias when only a constrained taxonomic section is examined. Web interaction studies have the advantage of being able to compare restored with reference sites, and also of allowing comprehensive field studies within a short time frame.

The long-term persistence of species within a habitat is more likely when restored or created habitats are managed for both their plants and pollinators (Dixon, 2009). It has been suggested that restoration may be much more successful if due attention is applied to pollinators in the early stages (Neal, 1998). The direct effects of pollination within a restored habitat may be quite apparent e.g. seed set. The indirect effects may be less apparent, for example, gene flow and genetic diversity, but none the less essential for the long-term success of restoration (Montalvo et al., 1997). Restored landfill sites may be valuable habitats for supporting flower visitors but pollinator interactions are also important as indicators of 'successful' restoration. This chapter attempts to answer this second issue, of determining the success of restoration on landfill sites, using field data to examine connectance and nestedness properties. Success in this context refers to the restored landfill sites having a similar extent of interaction linkages and nestedness as do reference sites.

Aims

The aim of this chapter is to determine how successful restored landfill sites have been in recreating the plant-flower visitor species interactions and structure found in natural habitats. The flower-insect interaction structure was compared for the restored landfill sites and the reference sites in relation to connectance and nestedness properties.

Connectance is the total number of realised linkages as a ratio of the total number possible within an interaction matrix. It is an inherent value useful in determining the functioning within a mutualistic assemblage of plants and pollinators, being a scale independent measure of the generalisation of a network (Jordano, 1987; Olesen and Jordano, 2002). The degree of connectance of a system co-varies with species richness, similarly to that of other food webs (Jordano, 1987). The expectation would be that the reference nature sites will have a stronger interaction pattern and hence connectance in terms of realised links between plants and flower visitors. Reasons for this may relate to being older habitats (Forup and Memmott, 2005b; Forup et al., 2008). Lower connectance values predicted for restored landfill sites, would suggest these habitats were in a greater state of fluctuation than the reference sites, possibly due to the restoration management (Jordano et al., 2003).

Nestedness is common in mutualistic networks such as assemblages of plants and pollinators (Bascompte et al., 2003; Dupont et al., 2003; Ollerton et al., 2003). A description and definition of nestedness is given later in the data analysis section of this chapter. In these networks, specialist plants or pollinators interact preferentially with generalist plants or pollinators (Bascompte and Jordano, 2007), and the generalist plants and pollinators will also interact with each other (Waser et al., 1996).

Studies of ecological networks can be prone to sampling effects as they involve entire species assemblages, some of which are less common or harder to sample. However it has been shown that nestedness analysis is less prone to bias from sampling effort than other forms of network analysis; Nielsen & Bascompte (2007) showed that the average absolute nestedness value does not change with sampling effort after an initial minimum amount of sampling. The value of the nestedness approach to interaction studies, compared for example to connectance analysis, is therefore that the stability of the absolute value occurs at a lower sampling effort (Nielsen and Bascompte, 2007). Within this study both nestedness and connectance are compared. Nestedness is a feature of binary or presence /absence matrices and is of particular relevance for studies focusing on the patterns of species occurrence among a set of locations and the pattern of interacting species within ecological networks (Almeida-Neto et al., 2007).

Determination of nestedness is important as it is a test of system robustness (tolerance to species extinctions). Studies on interaction webs have predicted robustness against extinctions at higher degrees of nestedness (Memmott et al., 2004; Burgos et al., 2007; Almeida-Neto et al., 2008). In nested interaction webs, the second most generalised pollinator interacts with a subset of the most generalised, and the third with a subset of the second and so on (Memmott et al., 2004). Nestedness also gains robustness from specialist flower visitors associating with generalist plants and vice versa, and so the loss of a plant or flower visitor species results in a linear loss of their connected species, as other generalised subset of connections remain (Memmott et al., 2004). However, interaction webs may not be robust to the simultaneous removal of the most generalised plants and flower visitor species (Memmott et al., 2004), and robustness may rely upon those species with fewer connections having a greater probability of extinction (Burgos et al., 2007). This study is taking an alternative perspective on the robustness and nestedness relationship as restoration relates to the return of species and hence the development of robustness within an interaction web. The expectation would be that longer established communities may have greater nestedness given that they may support greater abundance and richness of species. This would lead to greater interaction richness between plants and flower-visiting insects and hence show increased nestedness.

Methods

Study region and study sites

The study was conducted in the East Midlands of the UK, in the counties of Northamptonshire, Bedfordshire, Warwickshire and Buckinghamshire. All of the sites are within 30 miles of Northampton. Nine landfill sites were surveyed in the first year (2007) of the study accompanied by paired reference sites (See Chapter 2 - Table 2.01). The following year (2008), a random sub-set of three landfill sites accompanied by paired reference sites were surveyed. For further details on-site selection, see Chapter 2.

Fieldwork timing

Fieldwork surveys were conducted from March to October 2007 and 2008, as this corresponds to the main flowering period in central England and hence to flower visitor activity. Sites were sampled in random order with their paired references on consecutive days where possible. For distribution of survey days see Chapter 2-Table 2.02.

Flower visitor surveys

Flower visitor surveys were undertaken three times between 9am and 4pm on days which were warm and sunny with little or no wind, as outlined in the Butterfly Monitoring Scheme (Pollard and Yates, 1993) and similar to those used in previous pollination studies e.g. Goverde et al., 2002; Kleijn and van Langevelde, 2006; Potts et al., 2006; Nielsen and Bascompte, 2007 . Surveys each lasted 30 minutes and all flower visiting insects seen to be feeding legitimately (i.e. not nectar robbing) and large enough to touch anthers and stigmas were captured. For further details on methods see Chapters 2 & 4.

Data Analysis

The data were tested for normal distributions using one-sample Kolmogorov-Smirnov tests. Levene's test was used to determine whether variances were significantly homogenous, and the significance levels were adjusted accordingly if heterogeneous. For testing differences between the types of sites within the pairing, paired samples t-tests and Wilcoxon signed rank tests were used to compare parametric and non-parametric data, respectively. SPSS statistical software version 11.5 was used (SPSS, 2003). Abbreviations have been used: restored landfill and reference comparison-sites may be referred to as 'landfill' and 'reference' respectively. Significant results: $p \leq 0.05$.

Assessment of interaction structure

Interaction structures for flower visitors were drawn using Pajek software (Batagelj and Mrvar, 1998). Interaction matrices were assessed for interaction overlap and uniqueness between pairs of restored landfill and reference sites for 2007 and 2008.

Core species of plants and insects were identified. The core plant and insect species are defined here as those which are from the top 25% of species with the most interactions, and interacting with at least 25% of their bipartite species. These core plant and insect species typically interact with each other within an interaction web. Core species are different from keystone species in that they are important at facilitating functioning within an interaction web, but their individual loss is not critical to the web. Other species may take their place, and species may fluctuate from being core species or not. It is rather the relative importance of the species at a particular time. Removal experiments have determined with the loss of one core plant species, flower visitors readily move on to an alternative (Tarrant and Ollerton, Unpublished data).

Connectance analysis

Interaction studies on assemblages of plants and pollinators have examined the functioning of the habitat as the number of realised potential links between plants and their flower-visiting insect species (Forup and Memmott, 2005b).

Connectance is: **C = I/M**

Where **I** is the total number of realised interactions in the network, and **M** is the number of potential interactions in the network (number of plant species in flower x number of flower-visiting insect species) (Jordano, 1987). To remove the potential effect of those sites with more species having a higher value the interaction abundance is divided by the matrix size (Jordano, 1987; Forup et al., 2008).

N.B. For the calculation of connectance 'plant species in flower' refers to those plant species from within the interactions. For 2007 the interaction data collected for the year was collated to produce a single interaction web representing the plant-flower-insect interactions over the whole flowering season. For 2008, both seasonal and annually collated data were assessed.

Generalisation analysis

Generalisation within insect pollinated flowering plants and flower-visiting insects were determined using:

Connectance p (Plant generalisation) = I / p

Connectance i (Insect generalisation) = I / i

Where: I = the total number of realised interactions in the network, p = plant species richness, i = insect species richness. The analysis of generalisation within a community, gives an indication of the relative number of species of plants a flower visitor species interacts with and the number of flower visitor species a plant species interacts with (Forup and Memmott, 2005b). Analysis was made between connectance p and i, and for site types.

Nestedness analysis

The aims of nestedness metrics are to assess the extent a matrix of presence and absence deviates from perfect nestedness. In bipartite pollination networks, the columns of the matrix represent the plant species and the rows the flower visitor species. Perfectly nested pollination networks are those where the specialist flower visitors will use a perfect subset of the plants visited by the generalist flower visitors, and the more specialist plants are visited by a perfect subset of the insects visiting more generalist plants (Petanidou et al., 2008). Firstly, a metric is used to quantify the pattern of nestedness; secondly comparisons are made with appropriate null models to assess the significance of the metric nestedness value, and then inference is made as to the possible reason for the pattern of nestedness (Ulrich et al., 2009).

ANINHADO software was used, being based on the earlier Nestedness Temperature Calculator software (Atmar and Patterson, 1995), which was criticised for overestimating nestedness significance as it uses an equal probability null model (Fischer and Lindenmayer, 2002). ANINHADO computes two metrics of nestedness, the recently-proposed NODF (Almeida-Neto et al., 2008) and the more commonly used matrix Temperature (*T*) (Atmar and Patterson, 1995). NODF (an acronym for Nestedness metric based on Overlap and Decreasing Fill (Almeida-Neto et al., 2008)) has been advocated as it has greater theoretical consistence (Almeida-Neto et al., 2008). This is because NODF quantifies whether poorer assemblages of species constitute subsets of progressively richer ones and whether less frequent species are found in subsets of the sites where the most widespread occur. In comparison, the more usual metric that is used, Temperature (*T*), is analogous to physical disorder; it quantifies nestedness as the deviation from a nested pattern due to unexpected absence and presences, respectively, in more and less ''hospitable'' sites (Ulrich et al., 2009). However, it is unclear whether the Temperature metric should be used for interaction networks as there is no reason to weight presence and absence cells by their distance from the isocline, and NODF may therefore be more appropriate (Almeida-Neto et al., 2008). However, *T* is useful to compare to studies published to date. Therefore in this study NODF values has been used for analysis of nestedness, and T values in

Appendices 5 & 6, are available for comparison. The ANINHADO software also determines separate NODF figures for columns and rows. Theses figures represent the respective nestedness of plant and flower visiting insect species and are a representation of generalisation.

Null model tests potentially control for the influence of matrix size and shape, since the simulated random matrices used are all the same size and shape as the empirical matrix being tested (Gotelli and Graves, 1996). The significance of the nestedness temperatures was determined against 1000 runs of the Ce & Er null models in ANINHADO to compare the actual matrix with theoretical predictions (Atmar and Patterson, 1995; Guimarães and Guimarães, 2006). The Er model randomly assigns presences to any cell within the matrix. The Ce null model takes the percentage fill of the rows and the columns into consideration when calculating the probability of the presence within an individual cell. It is therefore the more biologically significant of the two models. The NODF value of the null models represents the mean nestedness with a random arrangement given the fixed values of matrix size and fill gained from the empirical matrix being tested. The probability figures generated represent the probability that the matrix would have this level of nestedness if it had been randomly generated as in the null models.

For 2007 the samples were combined into an annual interaction matrix for each of the sites; this was required due to the seasonal matrices being too small for analysis on their own, the drawback with this is that non-observable "forbidden" interactions would appear as zeros, instead of being absent from the matrix; i.e. a spring flowering plant could not interact with an autumn active flower visitor (Jordano et al., 2006). This led to the modification of the survey method in the following field season. For 2008, the data is separated to produce interaction matrices for different seasons.

Results

A total of 100 hours were spent sampling flower visitor interactions, with 64 in the first year on 18 sites and 36 in the second year on 6 sites. Therefore, on average each site received 3½ hours of surveying in the first year and 6 hours in the second. Over the two field seasons there were 942 flower-visitor interactions recorded, 317 the first year and 625 the second.

Analysis of interaction similarity

The specific interaction structures were assessed for similarity within the pairs of restored landfill and reference sites for 2007 (Table 5.01). This showed an average of only 1.2% of specific species-species interactions were shared between pairs of sites, whilst the pairs of sites shared means of 4.4% of plant species and 17.6% of flower-visiting insect species (Table 5.01). For 2008, this showed an average of 3.1% of specific species-species interactions were shared between pairs of sites, which is low given that 22.4% of plant species and 29.8% of insect species are shared between sites (Table 5.02).

Core structure

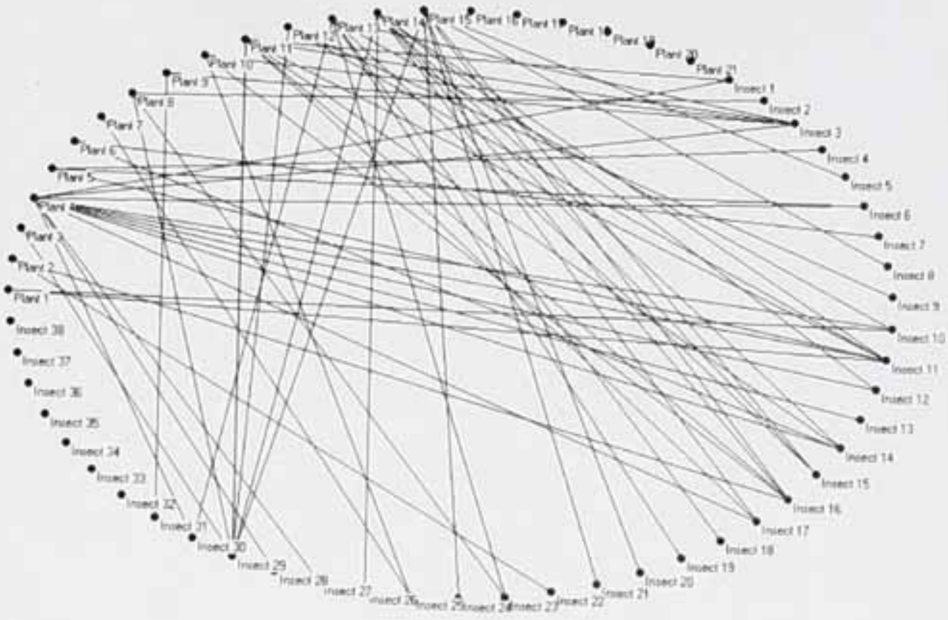
The core plant and insect species are defined here as those which are from the top 25% of species with the most interactions, and interacting with at least 25% of their bipartite species. Species interacting with only one other species were rejected. The Brixworth-Pitsford pair, shared one core plant and two core insect species, but the Sidegate Lane-Ditchford and Wootton-Barnes meadow pairs shared neither core plant nor insect species (Table 5.03 & 5.04 and Figures 5.01-5.03). Strong correlation was found between a species abundance and the number of interaction connections present for plants and insects on both restored landfill and reference sites (Figure 5.04).

Table 5.01 Similarity of richness of plant species in flower, insect visitors and flower-insect species interactions for pairs of restored landfill and reference sites 2007 (% of total for each richness variable). T– Combined number on both sites, B – Number found on both types of sites, RL – Number found only on the restored landfill sites, RF - Number found only on the reference sites (Plant and insect species richness from interaction matrix).

Pairs of sites	Plant species				Insect visitor species				Interaction richness			
Restored landfill - Reference	T	B	RL	RF	T	B	RL	RF	T	B	RL	RF
Bletchley – Blue Lagoon	9 (100)	0 (0.0)	5 (55.6)	4 (44.4)	14 (100)	1 (7.1)	8 (57.1)	5 (35.8)	20 (100)	0 (0.0)	11 (55.0)	9 (45.0)
Brixworth – Pitsford	15 (100)	0 (0.0)	8 (53.3)	7 (46.7)	27 (100)	3 (11.1)	14 (51.9)	10 (37.0)	39 (100)	0 (0.0)	25 (64.1)	14 (35.9)
Brogborough – Glebe Meadow	12 (100)	1 (8.3)	5 (41.7)	6 (50.0)	18 (100)	4 (22.2)	6 (33.3)	8 (44.4)	28 (100)	0 (0.0)	14 (50.0)	14 (50.0)
Cranford – Twywell	11 (100)	1 (9.1)	4 (36.4)	6 (54.5)	15 (100)	5 (33.3)	6 (40.0)	4 (26.7)	26 (100)	0 (0.0)	14 (53.8)	12 (46.2)
Harlestone – Scrub Fields	10 (100)	0 (0.0)	4 (40.0)	6 (60.0)	10 (100)	3 (30.0)	4 (40.0)	3 (30.0)	17 (100)	0 (0.0)	9 (52.9)	8 (47.1)
Kettering – River Ise Meadows	18 (100)	0 (0.0)	12 (66.7)	6 (33.3)	17 (100)	5 (29.4)	6 (35.3)	6 (35.3)	32 (100)	0 (0.0)	18 (56.3)	14 (43.7)
Kilsby - Draycote	4 (100)	0 (0.0)	1 (25.0)	3 (75.0)	7 (100)	0 (0.0)	1 (14.3)	6 (85.7)	9 (100)	0 (0.0)	1 (11.1)	8 (88.9)
Sidegate Lane – Ditchford	4 (100)	0 (0.0)	1 (25.0)	3 (75.0)	6 (100)	0 (0.0)	1 (16.7)	5 (83.3)	6 (100)	0 (0.0)	1 (16.7)	5 (83.3)
Wootton – Barnes Meadow	9 (100)	2 (22.2)	3 (33.3)	4 (44.4)	12 (100)	3 (25.0)	5 (41.7)	4 (33.3)	18 (100)	2 (11.2)	8 (44.4)	8 (44.4)
Mean percentage		**4.4**	**41.9**	**53.7**		**17.6**	**36.7**	**45.7**		**1.2**	**44.9**	**53.8**
Standard Deviation		7.7	14.2	14.1		13.2	14.2	22.5		3.7	18.4	18.7

Table 5.02 Similarity of richness of plant species in flower, insect visitors and flower-insect species interactions for pairs of restored landfill and reference sites 2008 (% of total for each richness variable). T– Combined number on both sites, B – Number found on both types of sites, RL – Number found only on the restored landfill sites, RF - Number found only on the reference sites (Plant and insect species richness from interaction matrix).

Pairs of sites	**Plant species**				**Insect visitor species**				**Interaction richness**			
Restored landfill - Reference	**T**	**B**	**RL**	**RF**	**T**	**B**	**RL**	**RF**	**T**	**B**	**RL**	**RF**
Brixworth – Pitsford	22	3	12	7	38	13	18	7	93	4	60	29
	(100)	(13.6)	(54.5)	(31.8)	(100)	(34.2)	(47.4)	(18.4)	(100)	(4.3)	(64.5)	(31.2)
Sidegate Lane – Ditchford	7	2	1	4	25	3	3	19	29	1	5	23
	(100)	(28.6)	(14.3)	(57.1)	(100)	(12.0)	(12.0)	(76.0)	(100)	(3.4)	(17.2)	(79.3)
Wootton – Barnes Meadow	12	3	5	4	30	13	11	6	60	1	38	21
	(100)	(25.0)	(41.7)	(33.3)	(100)	(43.3)	(36.7)	(20.0)	(100)	(1.7)	(63.3)	(35.0)
Mean percentage		**22.4**	**36.7**	**40.7**		**29.8**	**32.0**	**38.1**		**3.1**	**48.3**	**48.5**
Standard Deviation		7.8	20.5	14.9		16.1	18.2	32.8		1.3	27.0	26.7



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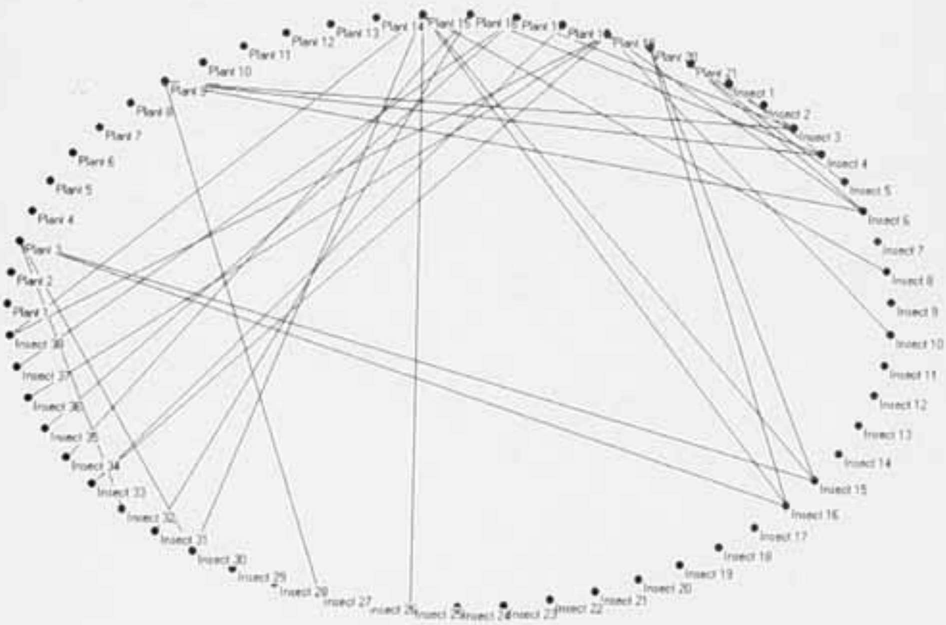
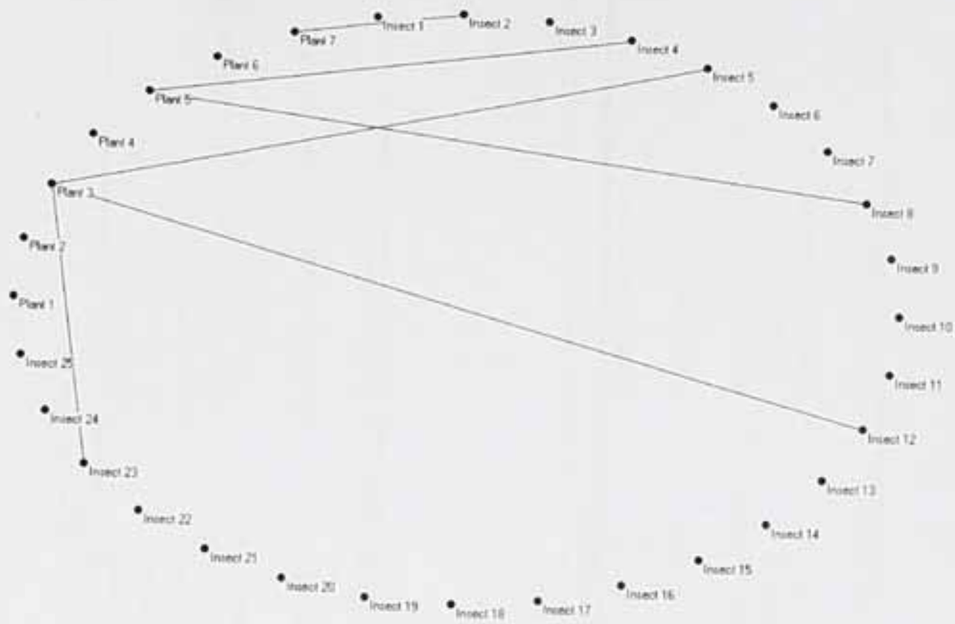


b)

Figure 5.01 Flower-insect visitor interaction structure in 2008 a) Brixworth restored landfill site, b) Pitsford reference site (Same plant and insect species identities were used for both interaction structures, where no link is present this refers to a species found on the comparison-sites, refer to Appendix 7 for species lists).



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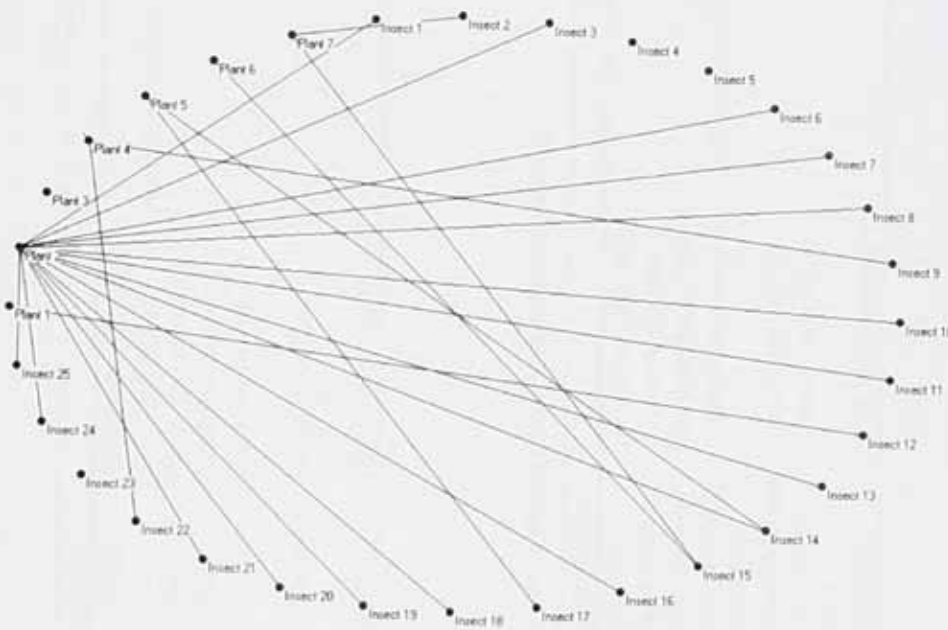


b)

Figure 5.03 Flower-insect visitor interaction structure in 2008 a) Sidegate lane restored landfill site, b) Ditchford reference site (Same plant and insect species identities were used for both interaction structures, where no link is present this refers to a species found on the comparison-sites, refer to Appendix 9 for species lists).

Table 5.03 Core plant and insect species for 2007 a) restored landfill, and b) reference sites.

NB. No core species present for Kilsby or Sidegate restored landfill sites.

****Core species found on both of the paired sites.**

a) Restored landfill sites

Site	Plant core species			Insect core species		
Bletchley	*Picris hieracioides*	*Taraxacum officinale*		*Apis mellifera***	*Calliopum* spp.	
Brixworth	*Cirsium arvense*	*Picris echioides*	*Picris hieracioides*	*Eristalis tenax*	*Calliopum* spp.	
Brogborough	*Picris echioides*	*Picris hieracioides*		*Apis mellifera***	*Eristalis tenax*	
Cranford	*Picris echioides*	*Picris hieracioides*		*Syrphus ribesii*	*Calliopum* spp.	*Apis mellifera*
Harlestone	*Cirsium vulgare*	*Trifolium repens*		*Bombus pascuorum***		
Kilsby						
Kettering	*Cardamine flexuosa*	*Senecio jacobaea*		*Meligethes aeneus*	*Bombus terrestris / lucorum*	
Sidegate Lane						
Wootton	*Cirsium arvense*	*Trifolium repens***		*Eriothrix rufomaculata*	*Ochlodes sylvanus*	

b) Reference sites (ordered to match paired restored landfill sites above).

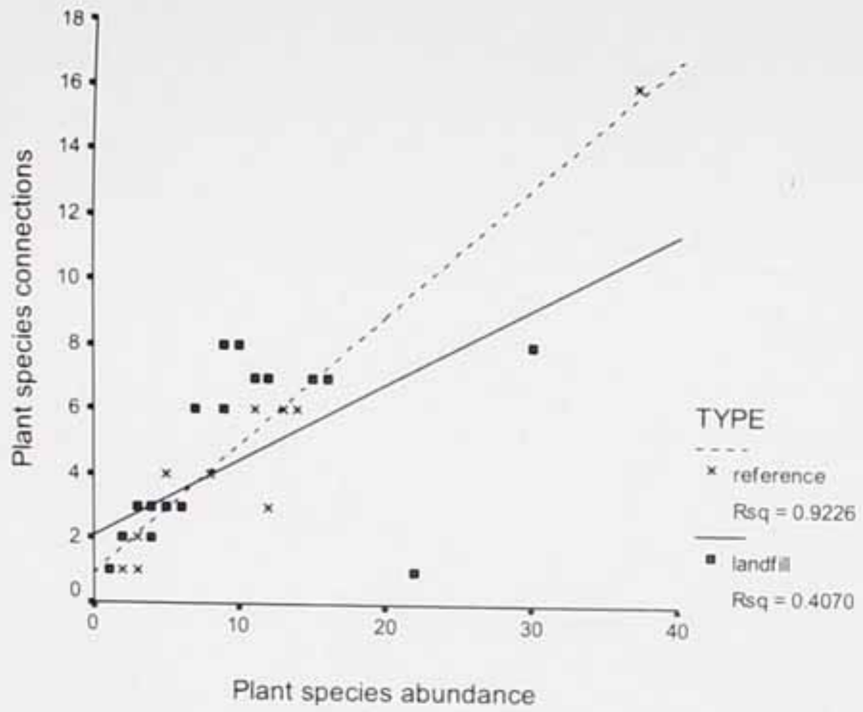
Site	Plant core species			Insect core species	
Blue Lagoon	*Centaurea nigra*	*Melilotus officinalis*		*Apis mellifera***	*Bombus pascuorum*
Pitsford	*Ranunculus acris*	*Stellaria giminea*		*Macropis europaea*	
Glebe Meadows	*Leontodon saxatils*	*Ranunculus bulbosus*		*Apis mellifera***	*Calliopum* spp.
Twywell	*Lotus corniculatus*	*Centaurea scabiosa*	*Trifolium dubium*		
Scrub Fields	*Centaurea nigra*			*Bombus pascuorum***	*Pipizini* sp.
Draycote	*Ranunculus bulbosus*	*Lotus corniculatus*		*Calliopum* spp.	*Apis mellifera*
River Ise Meadows	*Ranunculus ficaria*	*Centaurea scabiosa*		*Bombus pascuorum*	*Bombus lapidarius*
Ditchford	*Ranunculus acris*				
Barnes Meadow	*Picris echioides*	*Trifolium pratense***		*Bombus pascuorum*	*Syrphus ribesii*

Table 5.04 Core plant and insect species for restored landfill and reference sites 2008.

RL = Restored landfill site. RF = Reference sites.

****Core species shared on both sites in the pair**

	Brixworth - RL	Pitsford - RF	Sidegate - RL	Ditchford - RF	Wootton - RL	Barnes Meadow - RL
Plants	*Taraxacum officinale*** *Senecio jacobaea* *Cirsium arvense* *Picris hieracioides* *Ranunculus repens*	*Taraxacum officinale*** *Ranunculus acris* *Cardamine pratensis* *Lotus corniculatus*	*Ranunculus repens* *Taraxacum officinale*	*Ranunculus acris*	*Picris echioides* *Senecio jacobaea* *Trifolium repens*	*Ranunculus acris* *Ranunculus repens* *Trifolium pratense*
Insects	*Bombus lapidarius*** *Calliopum* spp. ** *Episyrphus balteatus* *Sphaerophoria scripta*	*Bombus lapidarius*** *Calliopum* spp. ** *Ferdinandea ruficornis* *Bombus terrestris / lucorum* *Bombus pascuorum*		*Macropis europaea* *Maniola jurtina*	*Eristalis cryptarum* *Eristalis tenax* *Ferdinandea ruficornis* *Sphaerophoria scripta*	*Anasimyia lineata* *Macrophya montana* *Pipizini* sp.



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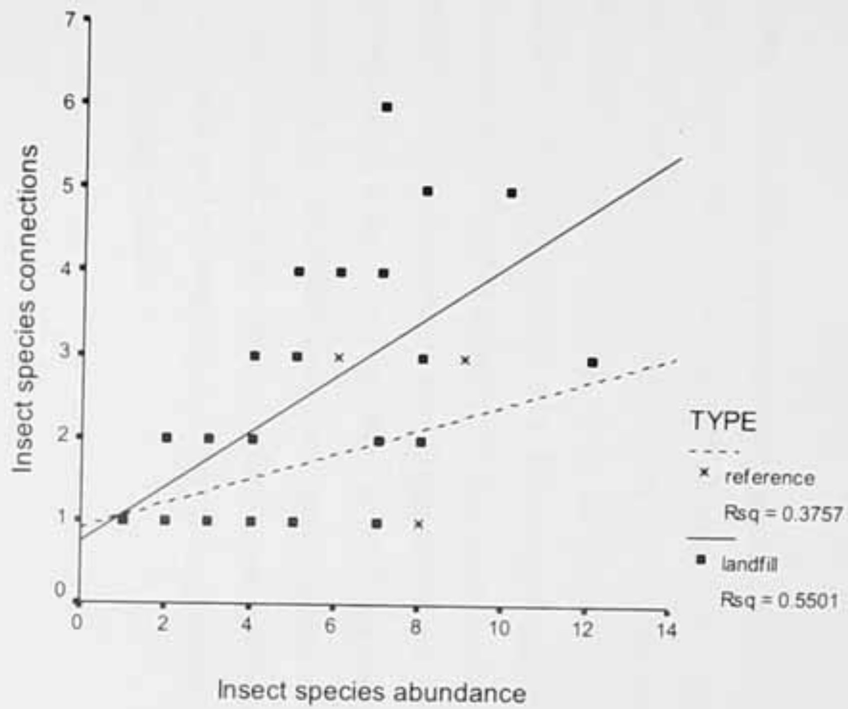
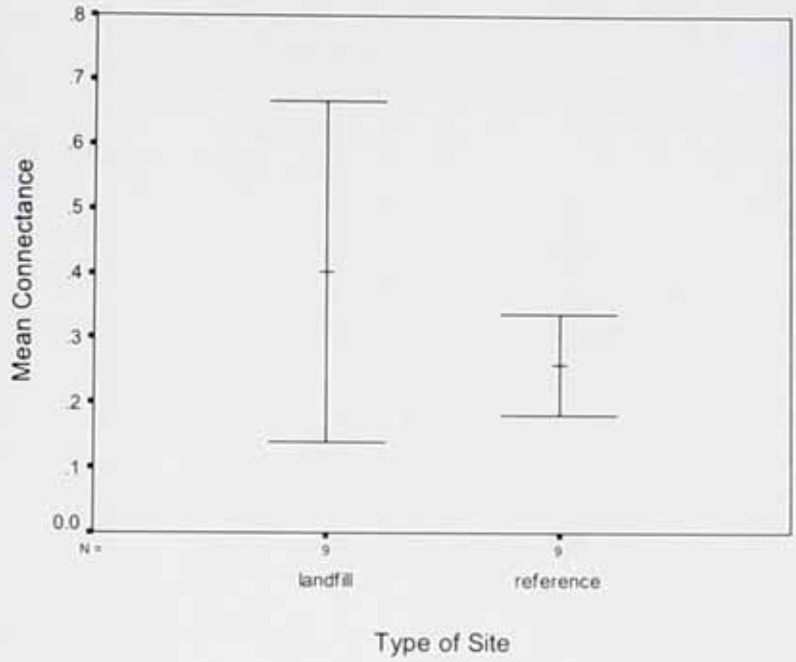


b)

Figure 5.04 Species abundance and number of interaction connections for restored landfill and reference site 2008: a) Plants. Spearman's correlation (Two-tailed); Restored landfill sites: (N=26) $r = 0.78$, $p<0.001$, reference sites: (N=22) $r = 0.91$, $p<0.001$. b) Insects. Spearman's correlation (Two-tailed); Restored landfill sites: (N=63) $r = 0.77$, $p<0.001$, reference sites: (N=62) $r = 0.58$, $p<0.001$.

Connectance

There was no difference in the mean annual connectance between the restored landfill and reference sites in 2007 or 2008 (Figures 5.05 a & b). There was greater variation observed for both years in the restored landfill data compared to the reference sites; Std. Dev. 2007: Restored landfills = 0.34, Reference = 0.10, Std. Dev. 2008: Restored landfills = 0.10, Reference = 0.01. For connectance on those sites surveyed both years, there was no difference between the two years for their mean connectance for either the restored landfill or reference sites (Figures 5.06 a & b). Each site was surveyed for an average of 3½ hours covering 700m^2 in 2007 and 6 hours covering 3600m^2 in 2008. Looking at the individual sites connectance, we see closely similar connectance values between years for most sites, with the exception of Sidegate Lane (Figure 5.07). This can be attributed to sampling on Sidegate Lane restored landfill site only recording a single interaction in 2007.



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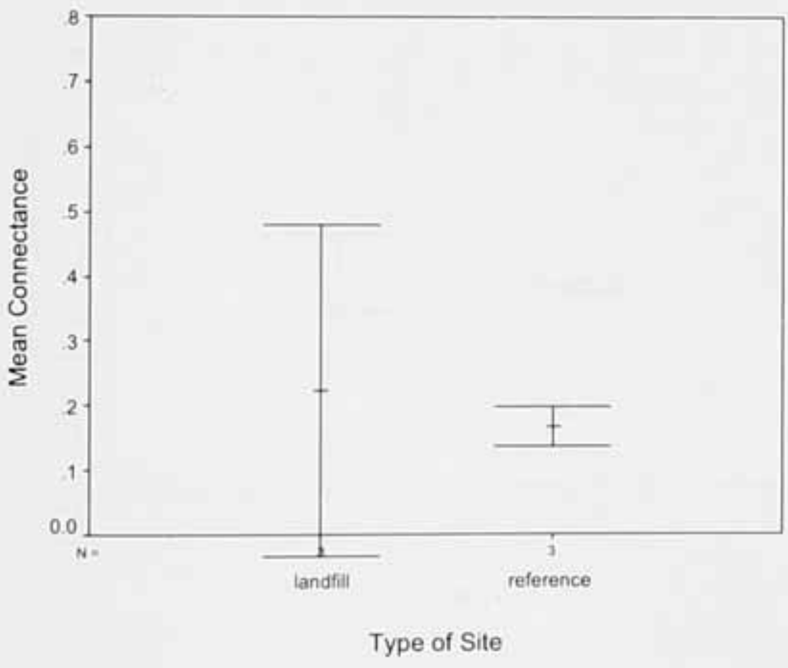
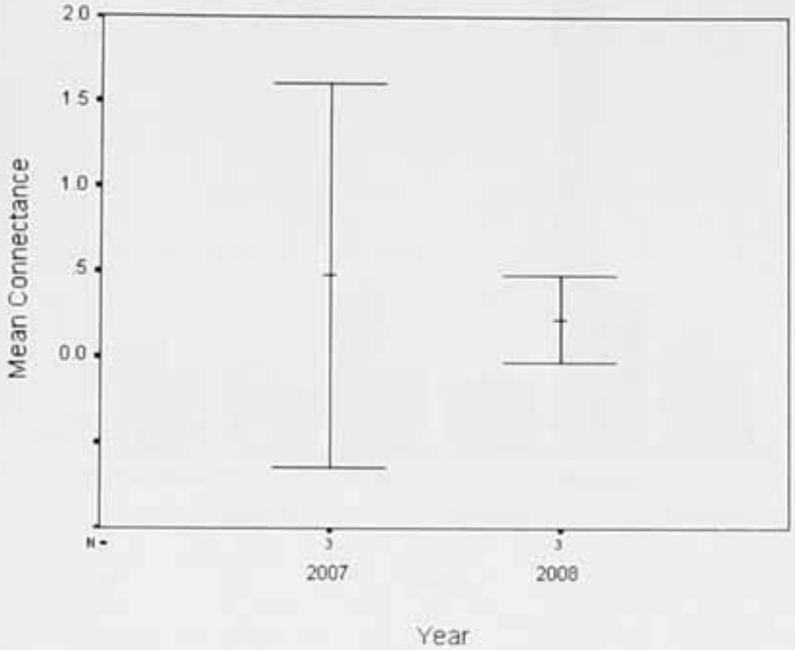


b)

Figure 5.05 Mean annual connectance (C) for restored landfill and reference sites (± 95% Confidence Limits) N=sample sizes. a) 2007 Paired samples t-test (two-tailed) t=1.55, df=8, p=0.16, b) 2008 Paired samples t-test (two-tailed) t=1.04, df=2, p=0.41.



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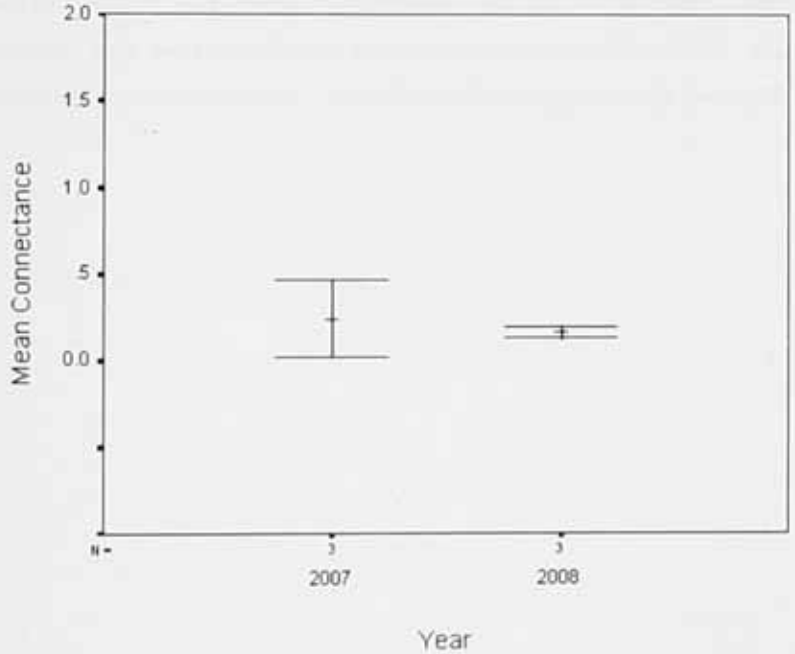


b)

Figure 5.06 Mean annual connectance for sites sampled in both years. (± 95% Confidence Limits) N=sample sizes. a) Restored landfill Paired samples t-test (two-tailed) $t=1.25$, $df=2$, $p=0.34$, b) reference sites Paired samples t-test (two-tailed) $t=1.66$, $df=2$, $p=0.24$.

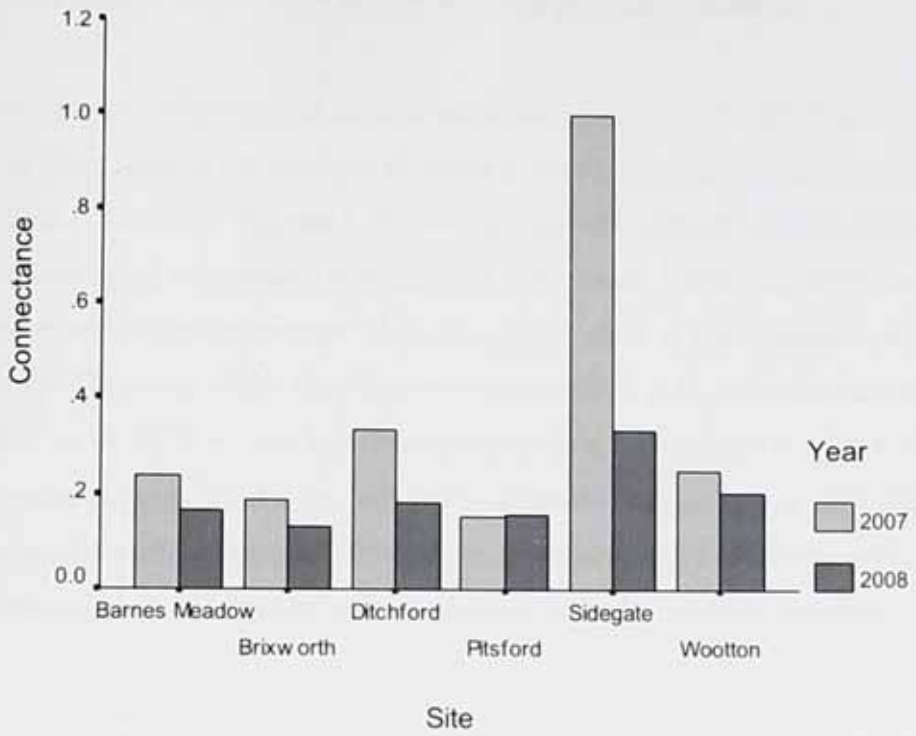
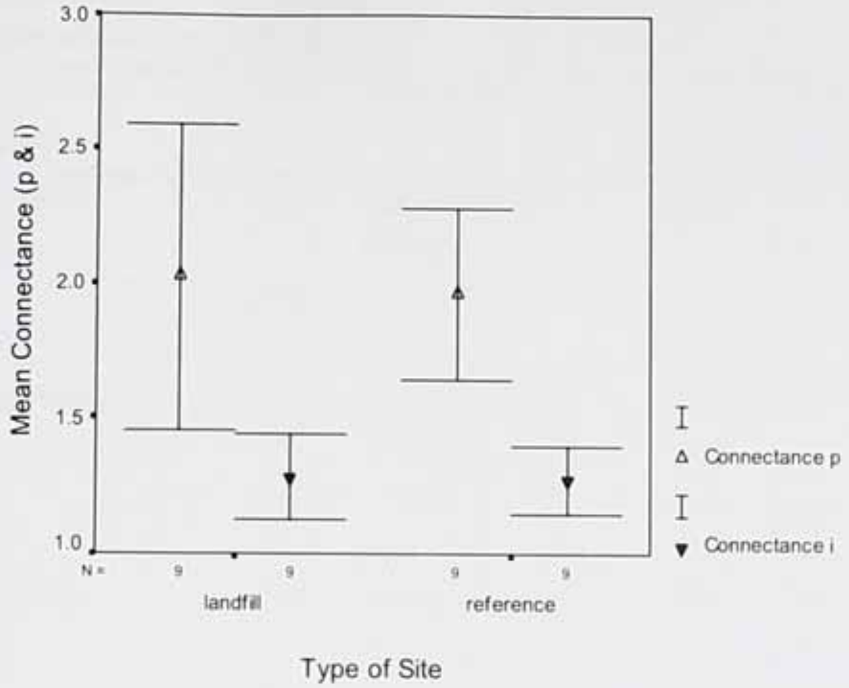


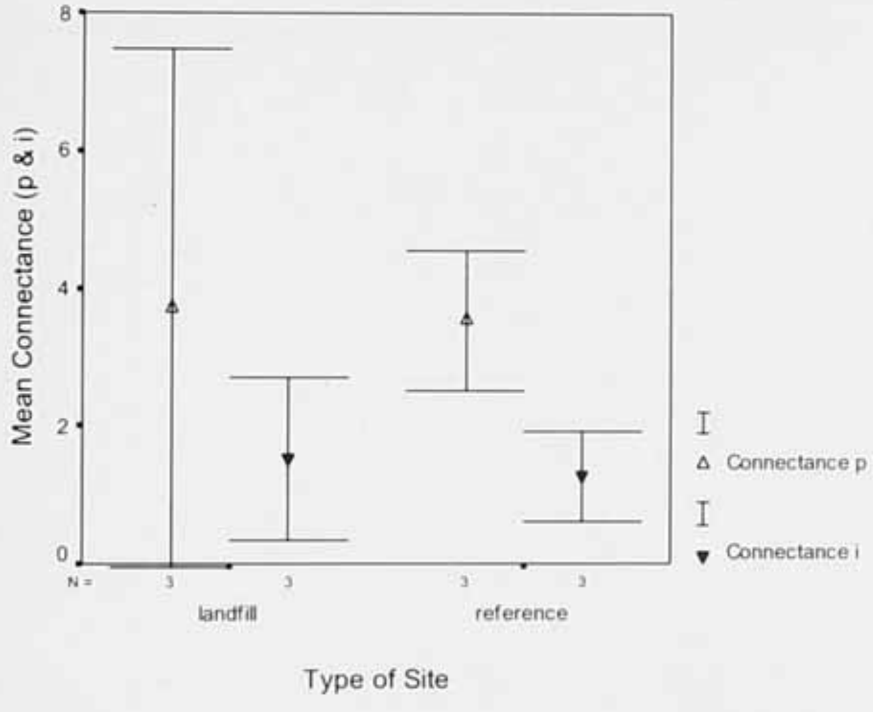
Figure 5.07 Annual connectance for sites sampled in 2007 and 2008. Restored landfill sites: Brixworth, Sidegate and Wootton. Reference sites: Barnes Meadow, Ditchford and Pitsford. NB. For Sidegate only one flower-insect interaction was recorded in 2007. Correlation of connectance values between years: Pearson's correlation (Two-tailed) r=0.96, p=0.002.

Generalisation of plants in flower and flower-visiting insects.

There was no difference for mean Connectance *p* (a measure of plant generalisation) or mean Connectance *i* (a measure of insect generalisation) for either types of site in 2007 or 2008 (Table 5.05, Figures 5.08 a & b). On both types of site in both years mean Connectance *p*, was greater than that of Connectance *I*, showing that plant species interact with a greater number of insect species, than do insect species with plant species. The mean of the ratio between Connectance *p* & *i*; for restored landfill sites 2007 = 1.56, 2008 = 2.40; and reference sites 2007 = 1.55, 2008 = 2.87. This shows that the ratios are very similar for both restored landfill and reference sites. Also the increased sampling effort in 2008 causes a proportionally greater increase in the number of insect species than plant species recorded in the interaction structure.



a)



b)

Figure 5.08 Mean Connectance for plants and insects on restored landfill and reference sites (± 95% Confidence Limits) N=sample sizes. Connectance *p* = plant generalisation, Connectance *i* = insect generalisation. a) 2007 Between site types; Connectance p: Paired samples t-test (two-tailed) t=0.12, df=8, p=0.91, Connectance i: Paired samples t-test (two-tailed) t=0.20, df=8, p=0.85. b) 2008 Between site types; Connectance p: Paired samples t-test (two-tailed) t=1.44, df=2, p=0.29, Connectance i: Paired samples t-test (two-tailed) t=0.17, df=2, p=0.88.

Table 5.05 Summary of Connectance values.
(Sample sizes for 2007: n = 9; for 2008: n = 3)

Connectance variable	Year	Type of site	Connectance			
			Mean	Min.	Max.	St. Dev.
Connectance	2007	Restored landfill	0.40	0.14	1.00	0.34
		Reference	0.26	0.15	0.44	0.10
	2008	Restored landfill	0.22	0.13	0.33	0.10
		Reference	0.17	0.16	0.18	0.01
Connectance $_p$	2007	Restored landfill	2.02	1.00	3.13	0.74
		Reference	1.96	1.33	2.67	0.41
	2008	Restored landfill	3.71	2.00	4.88	1.52
		Reference	3.53	3.29	4.00	0.41
Connectance $_i$	2007	Restored landfill	1.28	1.00	1.64	0.21
		Reference	1.27	1.00	1.50	0.16
	2008	Restored landfill	1.52	1.00	1.94	0.48
		Reference	1.27	1.09	1.57	0.26

Assessment of sampling completion.

An assessment of the extent of sampling completion was made, comparing the number of plant species found within the interaction matrix, with the number of plant species recorded in the floral surveys. For 2007, there was no significant relationship between the richness of flowering plants found in the interaction webs and the richness of flowering plant species found on-site (Figures 5.09): as a rough approximation one third of the plant species found on-site were found in the interaction matrices. The greater sampling in 2008 did give a tighter relationship, but this was only significant when both the restored landfill and reference site results were combined for correlation, with approximately one half of those plant species on-site were found in the interactions (Figure 5.10).

Given that both the restored landfill and reference sites had similar correlation between the plant species from within the interaction web and those from floral surveys; there are no perceived implications of this for the results gained for this study. The increased sampling effort in the second year did mean that a greater proportion of the interactions were recorded (Figure 5.10). The expectation is that that all insect pollinated plant species would receive flower-visitation.

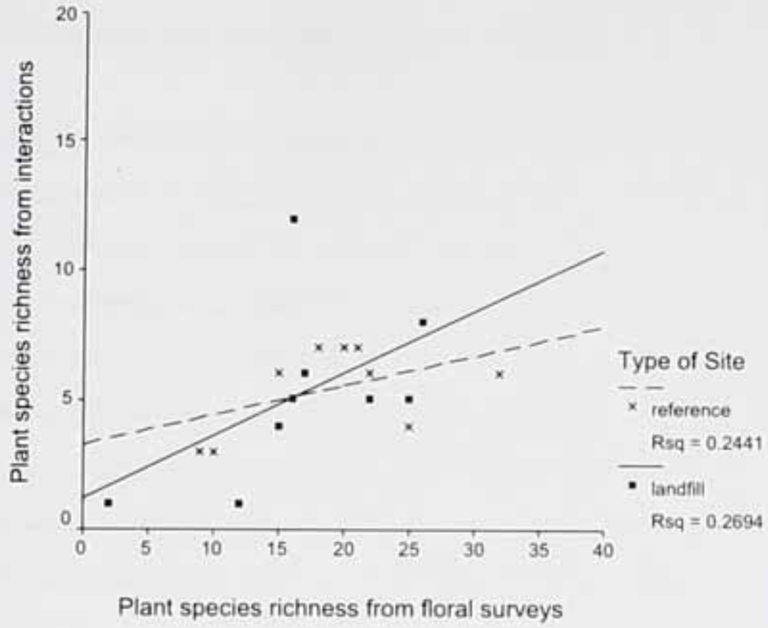


Figure 5.09 Richness of plants in flower from annual interaction webs and on-site annual richness of plants in flower for restored landfill and reference sites 2007. Pearson's correlation (Two-tailed); Restored landfill sites: (N=9) r = 0.52, p=0.15, reference sites: (N=9) r = 0.32, p=0.40.

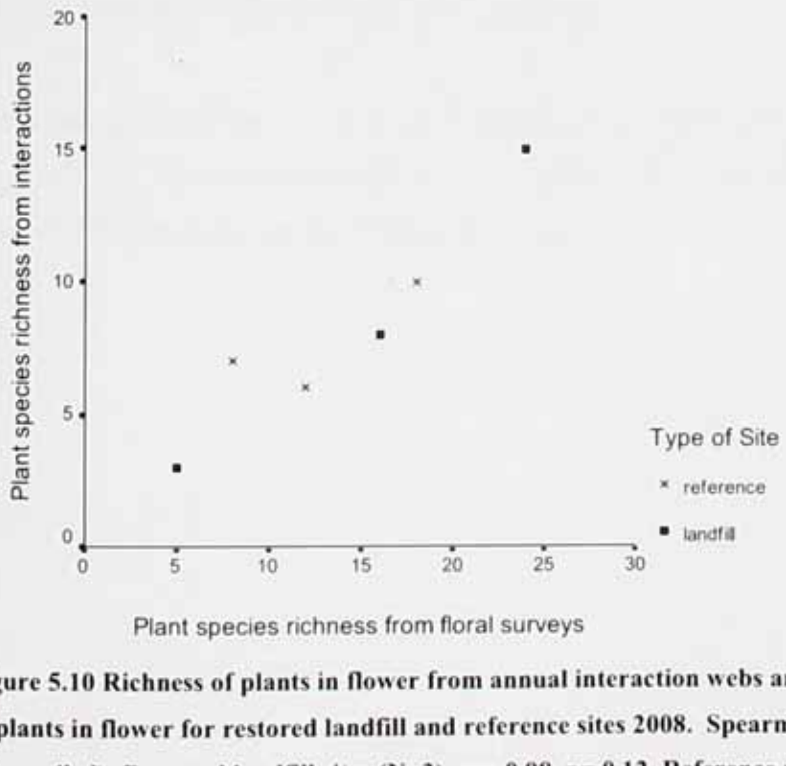


Figure 5.10 Richness of plants in flower from annual interaction webs and on-site annual richness of plants in flower for restored landfill and reference sites 2008. Spearman's rank correlation (Two-tailed); Restored landfill sites (N=3) r = 0.98, p= 0.12, Reference sites (N=3). r = 0.80, p= 0.42. Pearson's correlation (Two-tailed) for all sites: (N=6) r=0.94, p < 0.01.

Interaction matrix nestedness analysis

Examples of interaction matrices are given for Brixworth restored landfill and Pitsford reference sites as these sites were sampled both years and are a comparison pair (Figure 5.11). The nestedness metrics of the individual sites are presented in Tables 5.06 and 5.07, individual site nestedness was assessed for significance against two null models. The Ce and Er null models were run 1000 times in ANINHADO to find which interaction matrices were significantly nested (Tables 5.06 & 5.07) at $p \leq 0.05$. For 2007, no matrix was significantly nested. For 2008, only Brixworth restored landfill; summer and annual combined matrices were significantly nested. The results gained for both restored landfill and reference sites are therefore similar.

For the degree of nestedness for the plants and insects, there was a close correlation to that of the overall nestedness (Tables 5.06 & 5.07). There was no significant difference between the NODF values for the insects or plants for either restored landfill sites (Paired samples t-test (two-tailed) $t=-0.86$, $df=6$, $p=0.42$) and reference sites (Paired samples t-test (two-tailed) $t=0.88$, $df=8$, $p=0.40$).

There was no significant correlation found between NODF values and matrix size (Figure 5.12). This therefore supports the evidence of the robustness of NODF to species richness (Nielsen and Bascompte, 2007).

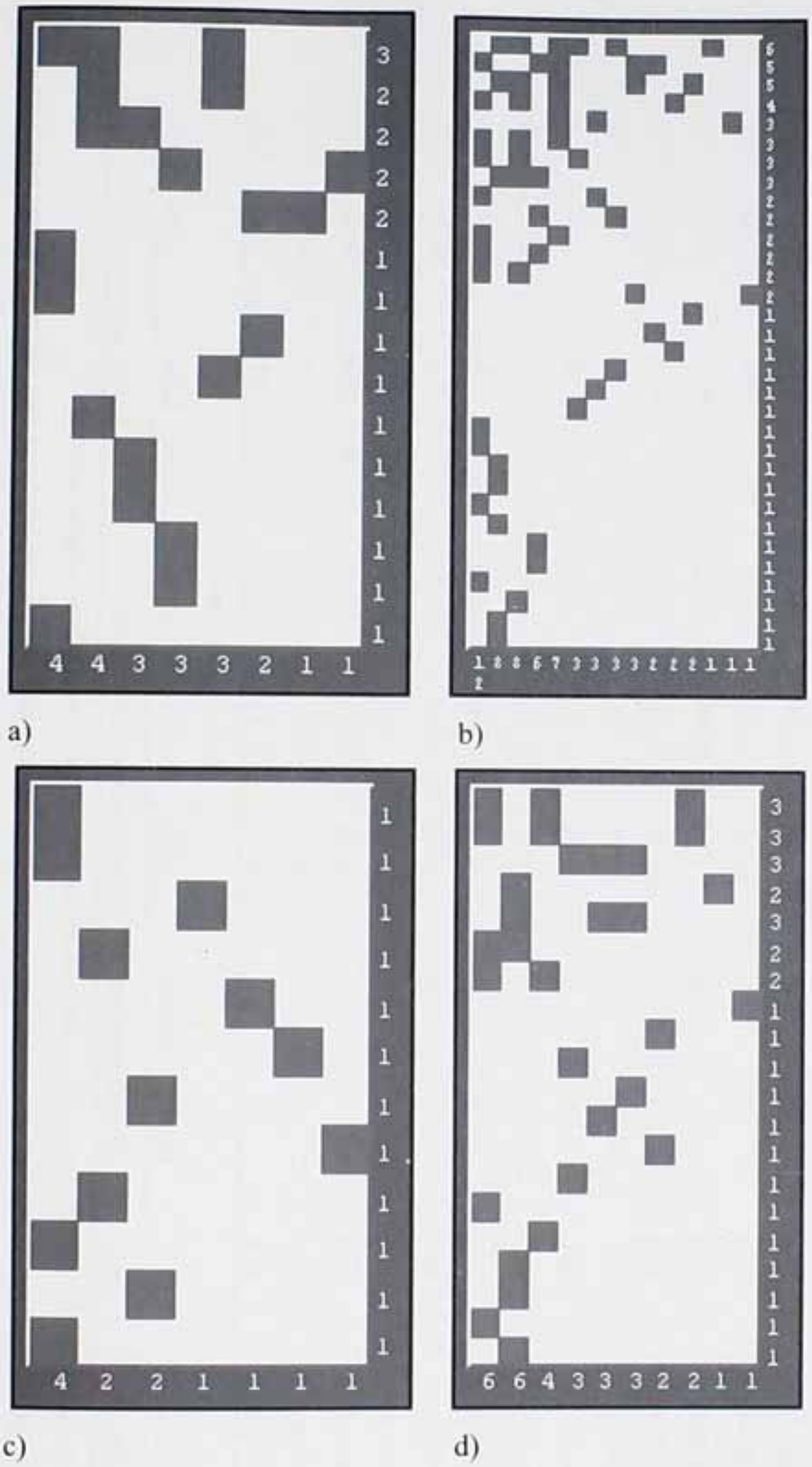


Figure 5.11 Examples of interaction binary matrices. a) Brixworth restored landfill site 2007, b) Brixworth restored landfill site 2008, c) Pitsford reference site 2007, and d) Pitsford SSSI reference site 2008. ■ indicates interaction recorded. Rows = insects, columns = plants. Numbers refer to interaction totals. Generated using Nestedness Calculator (Atmar and Patterson, 1995).

Table 5.06 Nestedness NODF and Null model analysis for flower- insect interactions on restored landfill and reference sites 2007. Table was sorted for site type and NODF Total. (Species richness refers to those within the interaction matrix, 'na' - matrix too small for analysis).

Site	Type	Fill %	Species richness		NODF degree of nestedness			NODF Null models			
			Plants (n)	Insects (n)	Total	Insects (rows)	Plants (columns)	(Er)	P(Er)	(Ce)	P(Ce)
Harlestone	Landfill	32.1	4	7	27.78	23.81	41.67	32.19	0.58	35.71	0.68
Brogborough	Landfill	23.3	6	10	24.58	22.22	31.67	24.42	0.47	28.32	0.63
Cranford	Landfill	27.3	5	11	21.03	21.82	16.67	25.69	0.68	29.24	0.79
Brixworth	Landfill	18.8	8	16	16.11	15.93	16.96	20.02	0.78	21.42	0.83
Bletchley	Landfill	24.4	5	9	15.22	16.67	10.00	24.54	0.79	25.69	0.82
Wootton	Landfill	25.0	5	8	14.47	14.29	15.00	25.42	0.83	26.44	0.84
Kettering	Landfill	13.6	12	11	11.16	10.00	12.12	14.73	0.75	16.05	0.81
Kilsby	Landfill	100.0	1	1	na	na	na	na	na	na	na
Sidegate Lane	Landfill	100.0	1	1	na	na	na	na	na	na	na
Blue Lagoon	Reference	37.5	4	6	47.62	46.67	50.00	37.32	0.24	42.30	0.37
Draycote	Reference	44.4	3	6	37.04	40.00	22.22	41.11	0.58	44.73	0.66
Barnes Meadow	Reference	23.8	6	7	25.00	26.19	23.33	25.13	0.45	27.60	0.54
Twywell	Reference	19.0	7	9	14.04	13.89	14.29	20.06	0.73	21.09	0.76
Glebe Meadow	Reference	16.7	7	12	10.63	12.12	5.95	17.36	0.85	18.83	0.87
Scrub Field	Reference	22.2	6	6	10.00	13.33	6.67	23.24	0.86	24.09	0.88
Pitsford	Reference	15.4	7	13	6.40	7.69	1.59	15.50	0.94	16.39	0.95
River Ise Mead.	Reference	21.2	6	11	2.86	0	13.33	21.91	0.99	23.41	0.99
Ditchford	Reference	33.3	3	5	0	0	0	31.14	0.84	33.07	0.85

Table 5.07 Nestedness NODF and Null model analysis for flower - insect interactions on restored landfill and reference sites 2008 (Species richness refers to those within the interaction matrix, 'na' - matrix too small for analysis).

Season	Site	Type	Fill (%)	Species richness		NODF degree of nestedness			NODF Null models			
				Plants (n)	Insects (n)	Total	Insects (rows)	Plants (columns)	(Er)	P(Er)	(Ce)	P(Ce)
Annual	Wootton	Landfill	20.3	8	24	22.09	21.80	24.91	22.07	0.49	24.45	0.68
Annual	Brixworth	Landfill	12.9	15	33	19.64	19.53	20.16	14.45	**0.02	17.08	0.18
Annual	Sidegate Lane	Landfill	33.3	3	6	0.00	0.00	0.00	31.43	0.89	29.50	0.86
Annual	Pitsford	Reference	15.7	10	21	15.56	16.19	12.59	17.14	0.65	18.67	0.76
Annual	Barnes Meadow	Reference	16.4	7	20	8.53	9.47	0.00	16.86	0.95	18.24	0.96
Annual	Ditchford	Reference	18.2	6	22	7.52	7.36	10.00	17.79	0.98	22.10	0.98
Spring	Wootton	Landfill	50.0	3	4	50.00	50.00	50.00	48.67	0.46	50.96	0.49
Spring	Brixworth	Landfill	50.0	2	10	0.00	0.00	0.00	37.75	0.95	36.18	0.93
Spring	Sidegate Lane	Landfill	100.0	1	2	na	na	na	na	na	na	na
Spring	Pitsford	Reference	52.4	3	7	52.08	47.62	83.33	50.17	0.49	52.95	0.56
Spring	Barnes Meadow	Reference	17.9	6	13	4.84	5.13	3.33	18.11	0.97	18.79	0.98
Spring	Ditchford	Reference	100.0	1	1	na	na	na	na	na	na	na
Summer	Brixworth	Landfill	13.1	13	24	20.80	20.89	20.45	14.59	**0.03	17.41	0.20
Summer	Wootton	Landfill	23.6	5	11	6.15	3.64	20.00	23.34	0.96	25.98	0.98
Summer	Sidegate Lane	Landfill	50.0	2	4	0.00	0.00	0.00	40.26	0.72	41.41	0.73
Summer	Pitsford	Reference	17.2	8	16	14.47	15.42	10.42	18.03	0.72	20.19	0.83
Summer	Barnes Meadow	Reference	22.5	5	8	9.21	10.71	5.00	23.05	0.88	24.00	0.90
Summer	Ditchford	Reference	21.0	5	21	0.91	0.48	10.00	20.29	1.00	24.10	1.00
Autumn	Brixworth	Landfill	33.3	5	9	47.28	50.00	37.50	34.88	0.13	39.31	0.25
Autumn	Wootton	Landfill	32.3	5	13	39.39	41.67	21.67	33.37	0.23	37.97	0.43
Autumn	Sidegate Lane	Landfill	0.0	0	0	na	na	na	na	na	na	na
Autumn	Barnes Meadow	Reference	0.0	0	0	na	na	na	na	na	na	na
Autumn	Ditchford	Reference	100.0	1	1	na	na	na	na	na	na	na
Autumn	Pitsford	Reference	0.0	0	0	na	na	na	na	na	na	na

** Significantly nested

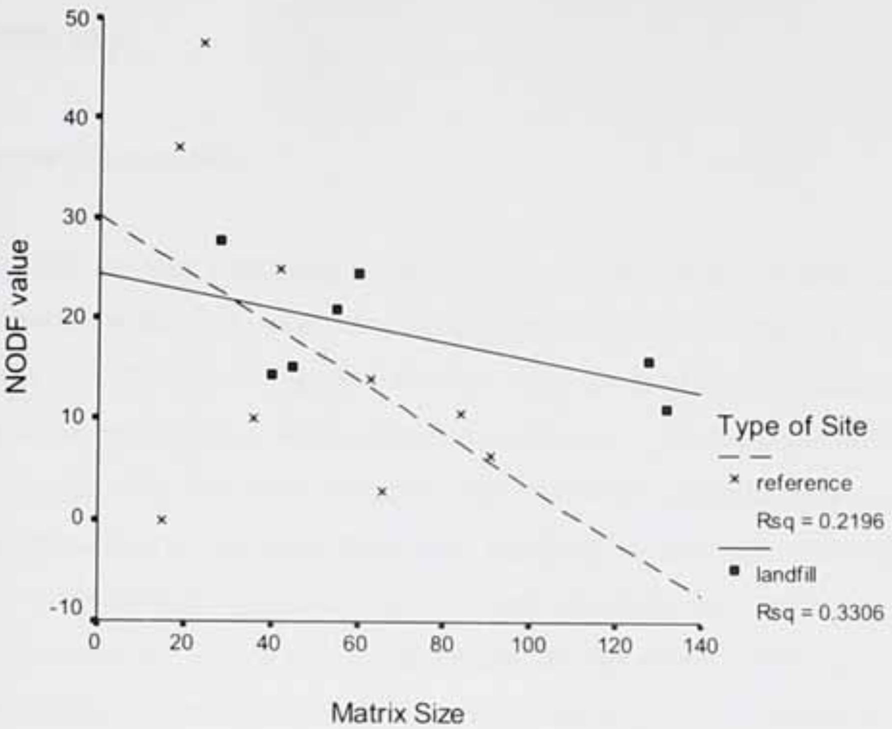


Figure 5.12 NODF annual values and matrix size for restored landfill and reference sites 2007. Pearson's correlation; Restored landfill sites: r=-0.58 p=0.18, Reference sites: r=-0.47, p=0.20.

Discussion

Interaction structure

A significant issue with using reference sites as targets to ascertain the success of restoration is that these sites often have different species of flora and fauna. From Tables 5.01 & 5.02 and Figures 5.01-5.03, the number of species shared within the interaction matrices on average range from 4% - 22 % of plant species and 18% - 30% of insect species. This gives weight to the argument for examining ecological processes and interactions to overcome these chief problems. In theory, flower-insect interactions are a comparable entity between restored and reference sites despite any variation in species structure. This is because they show the underlying ecological functioning of the sites. If plant-flower-insect interactions are occurring then the assumption is that pollination is occurring, and hence plant reproduction (Kevan and Baker, 1983). Plant reproductive output was not measured directly within this study given the limited resources; the use of bioassays for assessing pollination services in further work would be useful. Plant reproduction ensures the long term viability of a habitat, and supports other associated plant feeding animals including herbivores, frugivors and seed eaters. These supported organisms, in turn providing ecosystem services or food resources to other organisms. Arguably, pollination is the back bone ecosystem service allowing for the continuation of a habitat.

The restored landfill and reference sites were shown to have very low levels of interaction similarity. On average, only 1.2% - 3.1% of the plant-insect interactions were found on both pairs of sites (Tables 5.01 & 5.02). This is expected given the percentage of shared plants (e.g. plant spp. percentage shared x insect spp. percentage shared = expected percentage of shared interactions). Only those interactions between species present on both sites could therefore be recorded on both sites. For 2007, eight of the nine pairs of sites shared no specific interactions (Table 5.02). For 2008, Brixworth- Pitsford shared 4.3% with 4.65% expected; Sidegate – Ditchford shared 3.4%, with 3.4% expected; only the Wootton- Barnes Meadow shared a surprisingly low number of interactions of 1.7% with 10.8% expected (Table 5.02). Two of the three pairs of sites therefore have relatively high levels of shared interaction from those specific interactions possible. This low percentage of species and interaction similarity also gives weight to the use of metrics such as connectance and nestedness to assess ecosystem pollination functioning and hence restoration success.

Core species

Core species were determined for the restored landfill and reference sites (Tables 5.03 and 5.04). Most of the core floral resources were plant species with open flower types, e.g. *Taraxacum officinale, Ranunculus* spp. *Cirsium arvensis* and *Picris* spp. (Tables 5.03 & 5.04 and Figure 5.18). This is expected as these plants can provide floral resources to a wide range of flower-visiting insect groups, without them requiring specific tongue morphology for feeding. If these open types of flower are encouraged within habitat restoration then they are likely to encourage a broad range of flower-visiting insects, providing them with abundant available floral food resources. It has been previously shown that within plant-flower visitor assemblages the interactions are typically asymmetric, where generalist plant species support the more specialised insect species and generalist insects visit the more specialised plant species (Waser et al., 1996). It therefore may be that these generalist plant species on a site may support both the more common generalist and the rarer, more specialised flower-visiting insects.

For the core flower-visiting insect species, in 2007, the two commonest species were *Apis mellifera*, and *Bombus pascuorum* (Table 5.03). In 2008, the commonest species were *Bombus lapidarius*, *Calliopum* spp., *Sphaerophria scripta* and *Ferdinandea ruficornis* (Table 5.04). Core species of plants and flower visitors were not typically the same for the restored landfill and reference sites. For 2007, only one species of core plant was found shared on a pair of sites, *Trifolium repens* on Wootton and Barnes Meadow (Table 5.04). The core insect species were more commonly found on pairs of sites with *Apis mellifera* found on two pairs of sites, and *Bombus pascuorum* on one pair. For 2008, only on the Brixworth landfill – Pitsford reference site pairing were there any core species shared between sites (Table 5.04). Here, both sites shared *Taraxacum officinale* as a core plant and *Bombus lapidarius* and *Calliopum* spp. as core flower-visiting insects. No core species were shared between the other pairs of sites in 2008 (Table 5.04).

Of possibly more interest and value is whether sites have the same core species from one year to the next. This would show the relative importance of a particular species for the on-going ecosystem functioning of a site. For the six sites surveyed both years, four plants were found as core species in both years, one on each of four separate sites. Only two insect species were found in both years, one on each of two separate sites (Tables 5.03 & 5.04). Although this is a very short term study to make inferences from, given that more plant species were found as core species in both years than were insect species, this may indicate plant species at the core of interaction webs are less susceptible to population instability. More work is needed to determine the stability of the core species over the longer term, and during times of habitat disturbance.

The relationship between species connectance "coreness" and abundance was assessed (Figure 5.04). This relationship was found for both plants and insects, there was a significant correlation between their abundance and the number of connections they had with insects and plants, respectively. Whether this is a mathematical or statistical artefact is not clear, as abundance is linked to the number of connecting species i.e. with four connections there must be at least an abundance of four individuals recorded.

Mean connectance

There was no difference in the mean connectance found on restored landfill sites or reference sites in either 2007 or 2008 (Figures 5.05 a & b and Table 5.05). This indicates a similar level of species interaction between plants and flower-visiting insects. The restored landfill and reference sites' mean connectance values in 2007 and in 2008 can be compared to published values (Table 5.08).

The values for connectance on restored landfill sites from this study are quite high for 2007, compared to the other examples of plant-flower-insect connectance shown. The mean value for connectance on restored landfill and references sites are lower in 2008.

A similar relationship holds true for both years with restored landfill sites having greater connectance than the reference sites. Potential reasons for this relatively high level of connectance may be the low richness of plant species found within the interaction matrices on the sites. The negative correlation between connectivity and matrix size has been found previously, in a meta-analysis study (Spearmans rank correlation $r=-0.92$, $p<0.001$) (Olesen and Jordano, 2002).

The difference in connectance values observed between years is likely due to the specific connectance of those sites contributing to the subset of sites in 2008. The mean connectance values were determined for those restored landfill and reference sites which were sampled in both 2007 and 2008 (Figure 5.06). Both types of sites had the same mean connectance value in both years. This is interesting and shows that site connectance values may be constant for sites. For specific sites, there was significant correlation between values for both years (Figure 5.07). Also of interest is that the levels of connectance did not significantly change with an increase in sampling effort in the second year. This does raise further questions and areas of study relating to which site specific variables are therefore governing interaction web connectance, and whether this holds true over longer time.

Table 5.08 Examples of network connectance for plant-flower visitor insect systems.

System	Location	Connectance	Published
Restored landfill sites 2007	East Midlands UK	0.40*	**(This thesis)**
Reference sites 2007	East Midlands UK	0.26*	**(This thesis)**
Old Heathland	Devon, England	0.26*	(Forup et al., 2008)
Restored landfill sites 2008	East Midlands UK	0.22*	**(This thesis)**
Restored Heathland	Devon, England	0.19*	(Forup et al., 2008)
Reference sites 2008	East Midlands UK	0.17*	**(This thesis)**
Waste ground	Denmark	0.12	(Olensen, *unpub.*)[+]
Old hay meadows	SW England	0.08*	(Forup and Memmott, 2005b)
Grassland	Cass, New Zealand	0.06	(Primack, 1983)
Restored hay meadows	SW England	0.05*	(Forup and Memmott, 2005b)
Grassland-conifer forest.	Colorado - USA	0.04	(Waser et al., 1996)
Mediterranean shrubland.	SW Spain	0.01	(Herrera, 1985)[#]

*** - Mean given.** [#] cited in Jordano (1987); [+] cited in Olesen and Jordano (2002)

Generalisation of flowering plants and flower-visiting insects

The levels of generalisation of plants and insects were similar for both restored landfill and reference sites in 2007 and 2008 (Figures 5.08 a & b and Table 5.05). The plants had greater levels of generalised visitation than the flower-visiting insects, which has often been found in other studies (Bosch et al., 2009). One reason for this may be the greater number of flower visitor species than plant species typically found within studies of flower visitor interactions. Pollination network studies are typically based on flower observations, where insects seen to visit flowers and potentially pollinate are recorded or captured. The abundance usually follows a skewed distribution with a few abundant species and many rarer species. It is possible that those flower-visiting insect species which have a naturally low density within a habitat or landscape will be perceived as specialists owing to their infrequent observation. There were many more individual inflorescences than individual flower visitors, and also there are fewer species of plants than insect species, and so given equal sampling effort there is a greater likelihood that plants will record more species of insect visitation.

That a higher number of specialist insect interactions are observed is at odds with the accepted idea that generalisation is the rule (Insect species connectivity with plant species, for both restored and reference sites 2008: Mean = 2.04, Median = 2.00) (Figure 5.13). Possible reasons, on a local spatial or temporal scale, are that flower visitors may act as specialists visiting few plant species. Secondly is the aspect of incomplete sampling. A four year study showed a marked decrease in perceived specialists, when species-species interactions were collated over the longer time period (Petanidou et al., 2008).

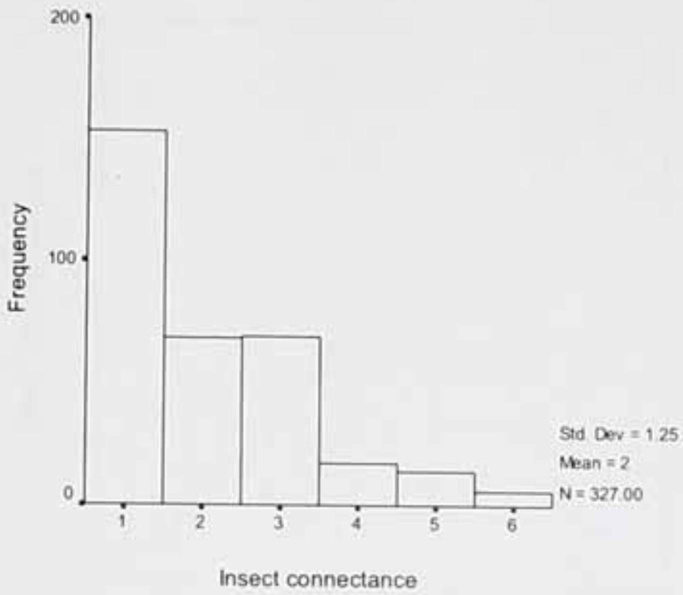


Figure 5.13 Frequency of insect species connectivity with plant species, for both restored and reference sites 2008.

Ratio for plants and insect species within interaction structure

The relationship between the number of plant and insect species within the interaction matrices was assessed. There were approximately twice as many insect species as plant species. A similar significant positive correlation was found for both restored landfill and reference sites (Figure 5.14). This relationship is similar to that found for Olesen and Jordano (2002) data set (Figure 5.15). This ratio between the number of plant and animal species within assemblages needs further examination and research, determining if this is possibly vegetative biome related and whether restricted to flower visitor mutualistic interactions.

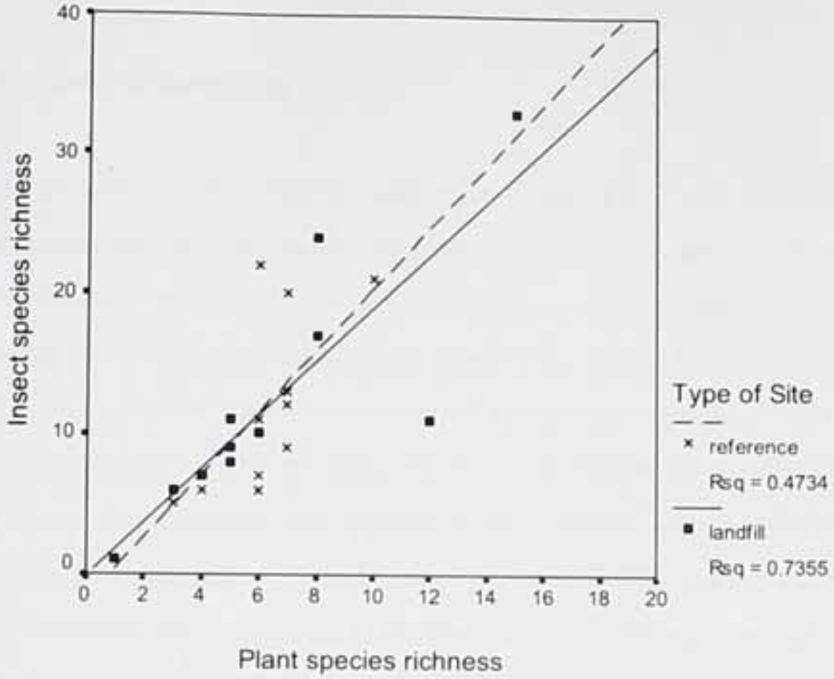


Figure 5.14 Plant and insect species richness for interaction matrices on restored landfill and reference sites. Both 2007 and 2008 are shown. Pearson's correlation (Two-tailed): Restored landfill sites: r = 0.86, p<0.001, reference sites: r=0.69, p=0.01.

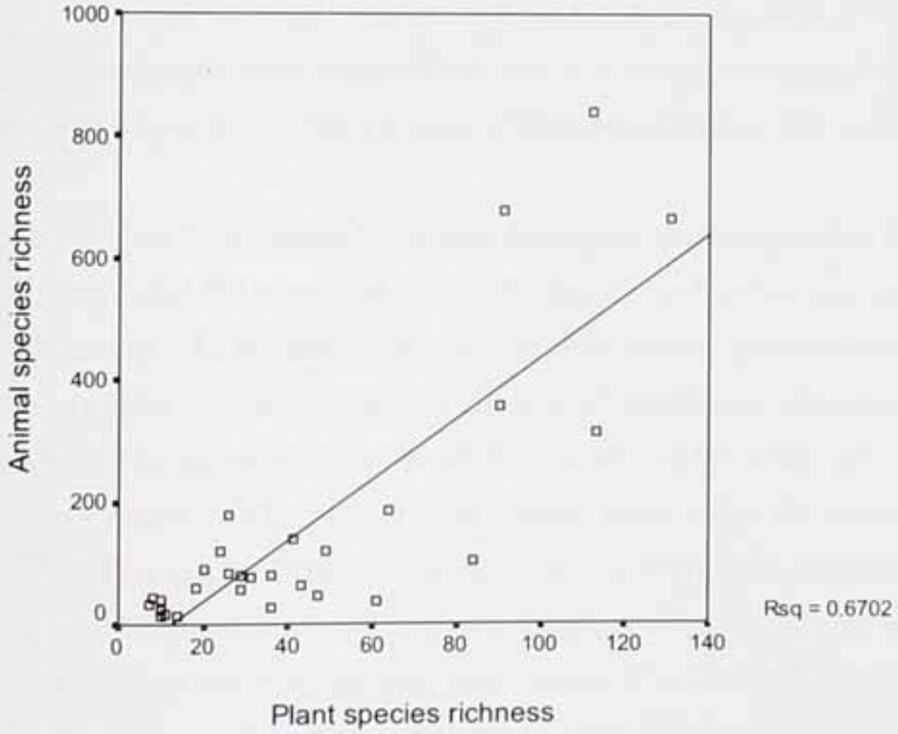


Figure 5.15 Examples of plant and animal species richness for interaction matrices. Data taken from Olesen and Jordano (2002). Pearson's correlation (Two-tailed): r=0.82, p<0.0001.

Assessment of sampling

In the assessment of sampling completion (Figures 5.09 & 5.10), there was no correlation between the number of plant species on-site and the number found within the interaction matrix. This may be indicative of incomplete sampling which may be missing the rarer interactions, as the expectation would be to see a significant correlation. Also in 2008, with increased sampling the proportion of plant species found within the interaction matrix from those on-site increased (Figure 5.10). However, as the incomplete sampling was present on both types of site, it will not significantly affect the comparison results. The richness of plants in flower used in calculating connectance did not necessarily include all of those present on-site and so values were correlated with the total floral richness found on-site. The relationship between the richness of flowering plants found within the interaction webs (this chapter) and the total number of flowering plant species on-site (Chapter 3) was assessed. The greater sampling in 2008 did give a tighter relationship, but this was still not significant (restored sites: 2007: Pearson's correlation $r=0.52$, $p=0.15$, 2008: spearman's correlation $r= 0.98$, $p = 0.12$). This is interesting as it was expected that sites with increased richness of plants in flower would have an increased number of flowering plants in their interaction web.

Further evidence of the disparity between the total richness of plants in flower and those found from within the interactions could represent the pollination services occurring on the habitat sites. The pollination services perceived here by flower-visitation rates, showed a similar relationship for both the restored landfill and reference sites. The proportion of the plants in flower found on the sites over the whole year was significantly more than the plant species richness found within the interaction matrix. For 2007 this was approximately a factor of three, and in 2008 a factor of two. Reasons for this could be attributed to the increased sampling effort. Further research is required into how common this is within other assemblages of plants and flower visitors. For annual plants this could be detrimental, but for perennials less so if they are pollinated and set seed in the following years. For 2007, the species richness of plants was recorded along the same transect as those used for the flower visitor interactions, whilst in 2008, the flower visitor interaction survey was across more of the site and so may have been just a function of the patchiness of vegetation.

This is not to say that the sampling regime was inadequate, as the surveying was applied evenly across both types of sites and therefore allowed for restored landfill and reference comparison. Of consideration was that there was no significant difference in the connectance values for those sites that were surveyed in both years (Figures 5.06 a & b). Each site was surveyed for an average of 3½ hours covering 700m^2 in 2007 and 6 hours covering 3600m^2 in 2008. This conversely, supports the suggestion that connectance is less susceptible to sampling effort than species richness (Martinez et al., 1999).

It has been described that connectance is robust to reductions in sampling effort (Vázquez and Aizen, 2006; Nielsen and Bascompte, 2007), however, it does remain to be seen whether there is a possible threshold to this relationship given extensive sampling (Figure 5.16). Following saturation sampling (the vertical dashed line), the speed of species accumulation may slow (A) and connectance may increase (B) rather than continue as expected (C). This would be due to the fill of the matrix increasing with increased sampling after the threshold saturation point. The saturation threshold point being where no new species are being recorded.

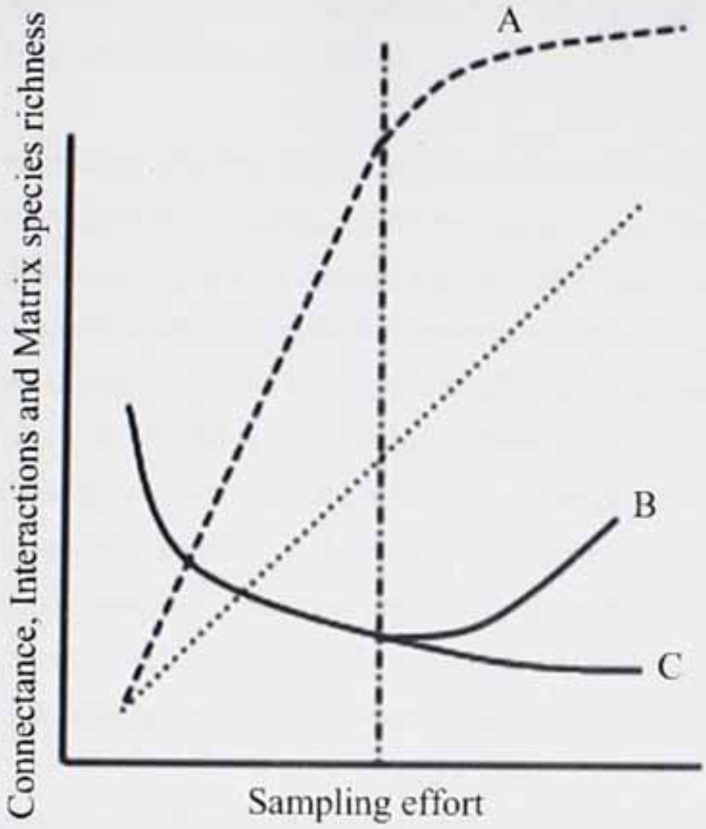


Figure 5.16 Theoretical sampling with relation to Connectance, Interaction abundance, Matrix species richness and Sampling effort. Solid line: Connectance, ------ = matrix species richness, ······· = interaction abundance. A – slowing increase in species richness, B – increasing connectance, C – continued rate of connectance. Vertical dashed line -.-.-.-.- = sampling saturation threshold.

Nestedness of interaction structure

The restored landfill sites had a mean nestedness of approximately 20 degrees, most empirical matrices have a 40-70 degree of nestedness (Almeida-Neto et al., 2008) (Tables 5.06 & 5.07). Few of the sites were significantly nested when compared to null models, none in 2007 and only Brixworth in 2008 (Tables 5.06 & 5.07). The exponential / logarithmic relationship between matrix size and nestedness suggests that only species rich systems will be highly nested, and small systems will not generally be nested (Bascompte et al., 2003; Guimaraes et al., 2006) (Figure 5.11). Nestedness is effectively undetectable below a particular threshold of species richness, a problem in describing small networks (Guimarães et al., 2005), although nestedness analysis is less sensitive to sampling effort than the number of species within the interaction structure (Nielsen and Bascompte, 2007).

Nestedness was therefore not significant for either restored landfill or reference sites, but is used here in comparing the two kinds of site. The aim for this research has been to determine whether the restored sites are functionally successfully restored, though comparison of their interaction structure with that found on reference sites. The maximum nestedness for restored landfill sites was lower than that for the reference sites (Tables 5.06 & 5.07). Increased nestedness has been suggested to increase robustness and decrease the effects of species extinction due to habitat loss (Fortuna and Bascompte, 2006). This therefore may indicate that reference sites are less robust than the reference sites, and may possibly be prone to extinction cascades following the loss of a plant or insect species from the sites. However, given that neither sites are significantly nested when compared to null models, this is difficult to conclude.

Ulrich et al. (2009) and Vasquez et al. (2009) highlight four possible ways that interactions may develop a nested structure, namely: i) individual abundance and passive sampling, ii) phenotypic matching, iii) plant phenology or temporal sampling, and iv) phylogenetic relationships. This study gives support to two of these reasons for nestedness. As previously described, the relationship between species connectance and abundance was assessed (Figure 5.04). This found for both plants and insects, there was a significant correlation between their abundance and the number of connections they had with insects and plants, respectively. Secondly this study supports how temporal sampling e.g. seasonally, and then collation into annual matrix, would promote subsets of nested species within the larger group. An example of this is shown in Figure 5.17, the interactions recorded on Brixworth restored landfill sites are clearly delineated by the months in which they were recorded. The interaction structure is being affected temporally, given the limited plant species overlap between seasons. This highlights the value of sampling across the whole year as phenologicaly, insects and flowers may not overlap. This also highlights the need within flower visitor conservation to find core floral resource species for each season, particularly within species poor habitats.

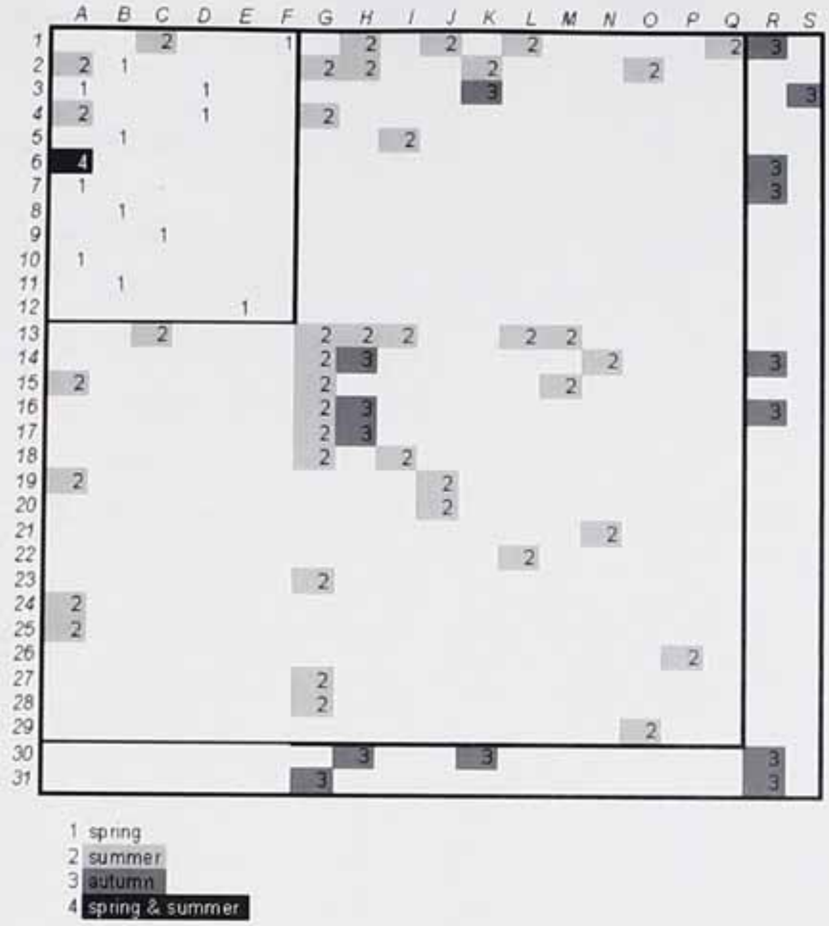


Figure 5.17 Interactions recorded over seasons for Brixworth restored landfill site 2008. X-axis *A-S* = plant species, Y-axis *1-31* = insect species. 1-4 seasons in which interaction were recorded. Border lines indicate area in which the majority of seasonal interactions occur. Phenological development can be imagined as running horizontally left to right.

In conclusion, the restored landfill sites have a similar interaction web as do the reference sites, regarding their mean connectance and nestedness. Pairs of restored landfill and reference sites share few species specific interactions, but this is inline with expectation given the proportion of species they share. The cores species found on the sites are generalists as expected and the more abundant species form more connections. Similar levels of plant and insect generalisation were found on both types of site, and plants interacted with more insect species than vice versa. Overall, the restored landfill sites can be determined as successfully restored given their similarity in interaction webs with the reference sites.

Results summary

In conclusion, the results of this part of the study can be summarised as follows:-

- Few plant and insects species were shared between pairs of restored landfill and reference sites and fewer specific interactions.
- In this study core plant species tend to be open flowered.
- The core species were the most abundant within the interaction structures in the study.
- Between years, sites had more core plant species shared, than core insect species.
- No significant difference was found between the mean connectance of restored landfill sites and reference sites in either 2007 or 2008.
- No significant difference was found in the mean connectedness values for those sites sampled in both 2007 and 2008.
- Plant generalisation was found to be greater than the insect generalisation for both restored landfill and reference sites in both years.
- No significant difference was found between the restored landfill or reference sites plant generalisation or insect generalisation found in 2007 or 2008.
- No significant correlation was found between the number of plant species found on-site or within the interaction structure.
- Neither restored landfill or reference sites were not found to be significantly nested and there was no significant difference found in the nestedness NODF values found on restored landfill or reference sites.

Figure 5.18 Core floral species with open flower types. Clockwise from top left: *Taraxacum officinale, Cirsium arvensis, Ranunculus* sp. and *Picris* sp.

Chapter 6

Conclusions and Recommendations

Conclusions and Recommendations

"There is nothing in which the birds differ more from man than the way in which they can build and yet leave a landscape as it was before."
Robert Lynd (1879 - 1949)

"Careful. We don't want to learn from this."
Bill Watterson (b.1958), "Calvin and Hobbes"

In this thesis the main objectives were to examine flowering plant species richness, floral resource abundance, flower-visiting insect richness and abundance and their interaction structure on restored landfill sites and to compare them with reference wildlife sites. This enabled determination of the restored sites' potential to support pollinating insects and the use of species interactions to indicate functionally successful restoration. No other studies have examined the assemblage of plants and their flower-visiting insects on restored landfill sites. This concluding chapter will consolidate and summarise the findings from this research and discuss the progress this has made in our current understanding of the potential for restored landfill sites to provide floral resources, support flower-visiting insects and determine success through examining species interaction webs. Additionally, recommendations are made for landfill site operators regarding restoration practice, and the limitations of this research are highlighted as are areas for future research.

The thesis was structured with Chapter 1 introducing the topic for study and the rationale behind this research project; and Chapter 2 describing the basic methods used. The main results for each of the other chapters will now be summarised and their implications discussed.

The examination of the floristic characteristics of restored landfill sites has showed us that they are providing as rich and abundant floral resources as do the reference sites.

There was no difference in the plant richness and floral abundance between restored landfill sites and reference sites. Approximately one quarter of plant species recorded in flower were unique to both restored landfill sites and reference sites, and one half were found on both types. The expected seasonal variation was apparent with floral resources and flower visitor activity peaking in the summer. In the spring the restored landfill sites had lower species richness and floral abundance than the reference sites. In the autumn the restored sites had higher species richness and floral abundance than the reference sites.

It has been seen within this study that the naturally revegetated restored landfill sites are well vegetated, and have floral resources equal to those sites which were sown. The most promising sites for natural colonisation are those with poor soils. Landfill site operators would benefit from using poorer soils as they are easier and cheaper to obtain for restoration than nutrient rich ones; often compost and soil 'improvers' are added to landfill top soils to meet environmental targets (Simmons, 1999; Watson and Hack, 2000). There is a cost benefit of using cheaper, poor soils and allowing natural revegetation, allowing significant savings to be realised. One possible consideration is whilst revegetation is awaited, soil run-off could occur; however with most opportunist species run-off would be minimal (Gilbert and Anderson, 1998). Where this is still of concern, possibly on those steeper slopes or sites, then the use of a grass nurse-crop of an appropriate species would be advantageous.

The policies brought in to regulate the restoration and after-use of landfills have had the unforeseen effect of creating well vegetated, florally diverse restored sites. The seed mixes, where used on sown sites, are cheap and grow readily on the soils, even though they may undergo difficult environmental conditions such as drought in the summer and water-logging in the winter (Gilbert and Anderson, 1998; Watson and Hack, 2000).

The long term after-use on the landfill sites may affect their continued floral resource provision. The sites studied were only up to 15 years old, and more research work needs to be done as the sites age. After they have become non-hazardous as designated by the Environment Agency, they are likely to be utilised for either agricultural or amenity purposes. This may however take 30 years or more (Watson and Hack, 2000). This change in landuse has yet to occur so it may be possible to intervene and persuade the landfill operators and policy makers that in England we are lacking in undisturbed restored sites which could be managed for the best possible benefit for flower-visiting insects, in so doing benefiting the landscape's biodiversity and proving valuable conservation habitats and seed banks for native species.

The decline of population of flower-visiting pollinating insects has been seen and attributed to habitat loss and degradation across the world (Allen-Wardell et al., 1998; Corbet, 2000; Steffan-Dewenter et al., 2002). Much of the research highlighting the current crisis relates to the decline of bee species (Williams, 1982; Westrich, 1996; Carvell, 2002; Goulson, 2003a; Ghazoul, 2005; Goulson et al., 2005; Carvell et al., 2006; McFrederick and LeBuhn, 2006; Inouye, 2007; Michener, 2007; Colla and Packer, 2008; Osborne et al., 2008a). However, their loss may be symptomatic of the loss of obscure none the less important pollinating insects which is going under reported; the available evidence suggests that many wild pollinators have declined dramatically in recent decades, in the UK, in Europe and globally (Buchmann and Nabhan, 1996; Kearns and Inouye, 1997; Corbet, 2000; Biesmeijer et al., 2006). The restoration of habitats which can support all flower-visiting insect groups is therefore of importance. For the comparison of flower-visiting insects both restored landfill and reference sites support rich and abundant flower visitor fauna. There was no difference between the restored landfill and reference sites mean flower-visiting insects' species richness or abundance. The same groups of insects were found on both types of sites, and there was significant overlap in species and abundance between the two types of site. Hoverflies, bumblebees and flies were the groups with the greatest species abundance on the restored landfill sites. Some groups of insects, namely hoverflies were more species rich and abundant on the restored landfill sites than on the references sites, whilst butterflies had less species richness and abundance on the restored landfill sites than reference sites. The most abundant species were common ones found on both types of sites.

Within the habitat restoration framework, not only should habitats be considered for their potential to support flower-visiting insects, but the flower-visiting insects should be encouraged to enable pollination of flowering plants (Dixon, 2009). It has been suggested that restoration may be much more successful if due attention is applied to pollinators in the early stages (Neal, 1998). The direct effects of pollination within a restored habitat may be quite apparent e.g. seed set. The indirect effects for example, gene flow and genetic diversity, may be less apparent, but are none the less essential for the long-term success of restoration.

The restored landfill sites have the potential to play a significant role in the conservation effort for the common flower-visiting insects now in decline. Notable loss of pollinator habitats in the UK include the loss of unimproved flower-rich grasslands (formerly valued as pasture and for hay production) (Goulson, 2003b) and the removal of hedgerows (Hannon and Sisk, 2009). The loss of habitats and semi-natural vegetation causes the loss of food resources, suitable nesting sites and materials. Flower visitors not only rely on habitats to supply their food sources, but also other life cycle requirements including nesting sites and materials. The decline of flower-visiting insect species has also been matched by losses in general biodiversity with parallel declines in butterflies, birds and plants on British farmland (Smart et al., 2000; Thomas et al., 2004).

The main issue with using reference sites as comparisons to ascertain the success of restoration processes is that they are different; this may be in space, soil type, but the principal problem is different vegetation structure. Research into the colonisation of new habitats has shown that new sites can vary in the species that become established from the available species pool, where ecological niches may be occupied by different species than on reference habitats (Simberloff and Wilson, 1969; Diamond, 1970; Schilthuizen, 2008). This is an issue in examining species with regards to restoration success, and highlights that functional and structural assessments are important. By examining an essential ecosystem service this has overcome these chief problems. The plant-flower-insect visitation web is comparable between restored and reference sites despite any variation in species structure. Ecological restoration and habitat creation is widely practiced, however the scientific determination of its success has been slow to develop. Species interactions can provide a good target for judging restoration success in comparison to species richness and abundance (Forup et al., 2008). The interaction network, characterises both functioning and species presence.

The restored landfill sites have a similar interaction web to the reference sites regarding their mean connectance and nestedness. The flower visitor interactions were analysed by their network attributes in order to verify structural and functional variation between restored landfill sites and reference nature sites of similar habitat type (See Chapter 5). There was no significant difference between the mean connectance of restored landfill sites and reference sites in either of the years in which fieldwork was undertaken. Pairs of restored landfill and reference sites shared few species-specific interactions, but this was in line with expectation given the proportion of species they share. The core species found on the sites were generalists as expected and the more abundant species form more connections. The core plant species visited by flower visitors were open flowered in morphology meaning that no specialist feeding morphology was required. Those species which were at the core of the interaction webs, forming the most connections, were also the most abundant. Similar levels of plant and insect generalisation were found on both types of site, and plants interacted with more insect species than *vice versa*. There was no significant difference in the nestedness found on restored landfill or reference sites. Overall, the restored landfill sites can be determined as successfully restored given their similarity in interaction web with the reference sites.

Restoration success

The Society for Ecological Restoration International - Science & Policy Working Group, provides a list of nine ecosystem attributes which can be used as a guide in measuring restoration success (SERI, 2004):

1. Similar diversity and community structure in comparison to reference sites;
2. Presence of indigenous species;
3. Presence of functional groups necessary for long-term stability;
4. Capacity of the physical environment to sustain ongoing populations;

5. Normal functioning;
6. Integration within the landscape;
7. Elimination of potential threats;
8. Resilience to natural disturbance;
9. Self-sustainability.

In the assessment of restoration success, in addition to evaluating the above criteria on site, it is necessary to compare the ecosystem attributes to those from reference sites (SERI, 2004). References sites should occur in the same biological region, within close proximity, and experiencing similar climatic and natural disturbances (Hobbs and Harris, 2001; SERI, 2004).

In practical terms most studies assess one or more of just three parameters; species diversity, vegetation structure, and ecological processes (Rhoades et al., 1998; Ruiz-Jaen and Mitchell Aide, 2005). Species diversity is usually measured by recording richness and abundance of species at different trophic levels (Nichols and Nichols, 2003). Vegetation structure is measured by determining the vegetation cover, biomass or vegetation profiles. Ecological processes such as nutrient cycling or species interactions are important as they indicate the functionality of a habitat. However, such processes are not considered as frequently as diversity and vegetation structure (Ruiz-Jaen and Mitchell Aide, 2005).

Successful restoration has often been seen in terms of species diversity rather than functioning processes. However, the simple presence of rare species, for example, is a blunt measure of success. True ecosystem function is seen in the species interactions, both mutualistic and antagonistic. Mutualism interactions are rarely considered in the restoration context; however they may be good indicators of a successfully functioning restored ecosystem. They show that species are present on the habitat sites and interacting and so demonstrating ecosystem functioning. The recovery of biological interactions in a restored habitat is critical for its long-term persistence (Ruiz-Jaen and Mitchell Aide, 2005). There has been a move towards recognizing that most ecosystems are dynamic and hence restoration goals and assessments cannot be based on static attributes (Hobbs and Harris, 2001). Therefore examining the species interactions is becoming recognized as of greater importance.

Within this thesis a species interactions approach has been applied to the question of restoration success. Within restoration ecology there needs to be a shift away from individual species, towards community interactions, and hence ecosystem functioning. The advantages of using flower-insect visiting interactions for determining restoration success, relates to this measure examining two trophic levels of organisms and also ecosystem functioning. The disadvantage is that it requires multiple measurements across the flowering seasons, which increase the time and so the cost of assessment.

Assessment of landfill site restoration success

The restored landfill sites considered in this study were assessed for successful restoration, in comparison to averaged results from the reference sites (Tables 6.01 & 6.02). Consideration was made as to whether the restored landfill sites should be compared against the lowest reference site values, but it is the opinion of the author that this would be underselling the potential of the restoration. The reference sites were wildlife sites, but showed variation within their provision of floral resources and potential to support flower-visiting insects. Therefore the mean result of their overall potential was a better target than the lowest of their floral resources or richness and abundance of flower visitors supported. Comparison with the single best reference is an unrealistic proposal.

The criteria and points system which were determined for governing successful restoration within this thesis for the sites assessed are as follows:

- A category would be deemed successful for a restored landfill site if it was above the mean of the reference sites (Scored 3 points).
- Site variables could also be given an intermediate pass if the variable value was below but within one standard deviation of the mean (Scored 1 point).
- Any values greater than one standard deviation below the reference mean were automatically deemed to be unsuccessfully restored (Scored 0 points).

- Greater importance was placed on flower-visiting insects rather than plants, as the focus for this study was the potential of the restored landfill sites for supporting *pollinating insects*. Therefore the scores for success for the variables of flower-visiting insects richness and abundance were weighted at one and a half times.
- Nestedness and connectance were examined independently against the mean from the reference sites.
- Sites can be deemed: successfully restored (scored ≥ 16); currently unsuccessful, but not critical (scored 8 <16; or unsuccessful and remediation required (scored ≤ 8).

Of the nine restored landfill sites surveyed in 2007, two were deemed as successfully restored, five were deemed unsuccessful but not critical, and two were deemed as unsuccessful and in need of remediation attention (Table 6.01). For 2008, two of the three sites surveyed were deemed successfully restored and the third unsuccessful (Table 6.02). Three restored landfill sites were surveyed in both 2007 and 2008, namely: Brixworth, Sidegate and Wootton. For 2008 the sites had greater surveying intensity than in the proceeding year. Two of the sites, Brixworth and Sidegate Lane, had the same result in both years, successful and unsuccessful respectively. Wootton was deemed as currently unsuccessful in 2007 and then as successful in 2008. This change in status is likely due to the increased sampling rather than a change in biodiversity or ecosystem functioning in such a short space of time. This change in categorisation for Wootton does highlight that caution should be applied in using this approach, although limited sampling will always exist to some extent.

The success of restored landfill sites to support pollinating insects may also be examined exclusively. Eight of the nine restored sites are below the reference site mean for richness of flower-visiting insects, whilst an equal number are above the reference site mean for flowering plant richness (Table 6.01).

The apparent successful return of flowering plants for the majority of sites may relate to early successional communities. Restored sites may quickly regain species richness of plants, but not of the same species as those found on longer established reference sites. The relatively unsuccessful return of flower-visiting insect richness may relate to insect species dispersal abilities. Although insects have greater mobility as individuals than plants, they still require both actual arrival and site specific requirements to become established.

Using both nestedness and connectance values in determining restoration success is useful. This is particularly true when interaction matrices are small. Where only a few interactions are recorded, this leads to a high connectance value, given the relatively higher fill of the matrix. Nestedness is common in mutualistic systems but not always present in species poor systems (Guimaraes et al., 2006). Comparing the relative levels of both connectance and nestedness may be important. In the early stages there may be higher connectance and lower nestedness, whilst older more established systems conversely may have lower connectance and higher nestedness. This relates to the balance between species richness of plants and flower-visiting insects and the number of interactions. Those sites which have been newly colonised by plant species, will take a while for the development of floral resources. Once this has developed then sites will begin to be visited by flower-visiting insect species. With relatively low abundance of species on-site at the start of restoration, high levels of flower-insect connectance will be observed, but it is unlikely that the assemblages will be nested, rather insect species being sampled predominantly from separate plant species. The habitats then continue to develop and the number of species builds up, here a decrease in the level of connectance is seen, as each new sample often adds new species to the matrix. The habitats continue to develop and abundance of individuals within species will increase. It is only following a build up in the abundance of insects of individual species that samples of insects will be taken from numerous plant species. This will then increase the probability that an assemblage of plants and flower visitors is nested. This balance between connectance and nestedness could therefore be used to show the stage of ecological development of a restored habitat.

Table 6.01 Summary of restoration success for restored landfill sites 2007 (values are compared to reference site means).

Restored landfill site	Richness of Insect-pollinated plants	Peak floral abundance (cm^2 per m^2)	Richness of flower-visiting insects	Abundance of flower-visiting insects	Interaction connectance	Interaction nestedness (NODF)	Points total	Successful restoration?
Bletchley	22 ✓	1.02 ×	9 ⭳	13 ✓	0.24 ✓	15.22 ⭳	**13**	☞
Brixworth	26 ✓	2.44 ⭳	17 ✓	31 ✓	0.18 ⭳	16.11 ✓	**17**	♁
Brogborough	17 ⭳	4.68 ⭳	10 ⭳	16 ✓	0.23 ⭳	24.60 ✓	**12**	☞
Cranford	25 ✓	2.81 ⭳	11 ⭳	13 ✓	0.25 ✓	21.03 ✓	**16**	♁
Harlestone	15 ⭳	14.31 ✓	7 ⭳	9 ⭳	0.32 ✓	27.78 ✓	**13**	☞
Kettering	16 ⭳	22.07 ✓	11 ⭳	20 ✓	0.14 ⭳	11.16 ⭳	**12**	☞
Kilsby	2 ×	0.01 ×	1 ×	1 ×	1.00 ✓	0.00 ×	**3**	♀
Sidegate Lane	12 ⭳	5.12 ⭳	1 ×	1 ×	1.00 ✓	0.00 ×	**5**	♀
Wootton	16 ⭳	7.10 ⭳	8 ⭳	12 ⭳	0.25 ✓	14.50 ⭳	**9**	☞

Reference means per site: 2007

Total richness of Insect-pollinated plants	= 17.50	St. Dev. = 7.14
Peak floral abundance	= 10.08	St. Dev. = 7.94
Richness of flower-visiting insects	= 11.50	St. Dev. = 6.29
Total abundance of flower-visiting insects	= 12.56	St. Dev. = 4.16
Annual interaction connectance	= 0.24	St. Dev. = 0.10
Annual interaction nestedness (NODF)	= 15.43	St. Dev. =14.26

✓ - Same or greater than reference mean.
⭳ - Below reference mean, but within 1 St. Dev.
× - Greater than 1 St. Dev. below the mean.
♁ - Restoration successful (scored ≥ 16).
☞ - Restoration currently unsuccessful, but no remediation required (scored 8 <16).
♀ - Restoration unsuccessful. Remediation required (scored ≤ 8).

Table 6.02 Summary of restoration success for restored landfill sites 2008 (values are compared to reference site means).

Restored landfill site	Richness of Insect-pollinated plants	Peak floral abundance (cm^2 per m^2)	Richness of flower-visiting insects	Abundance of flower-visiting insects	Interaction connectance	Interaction nestedness (NODF)	Points total	Successful restoration?
Brixworth	24 ✓	5.05 ↓	33 ✓	214 ✓	0.13 ✗	19.64 ✓	**16**	👍
Sidegate Lane	5 ✗	0.17 ↓	6 ✗	7 ✗	0.33 ✓	0.00 ✗	**4**	👎
Wootton	16 ✓	7.10 ✓	24 ✓	161 ✓	0.20 ✓	22.1 ✓	**21**	👍

Reference means per site: 2008

Total richness of Insect-pollinated plants	= 12.67	St. Dev. = 5.03
Peak floral abundance	= 5.80	St. Dev. = 5.87
Richness of flower-visiting insects	= 21.00	St. Dev. = 1.00
Total abundance of flower-visiting insects	= 58.00	St. Dev. = 27.62
Annual interaction connectance	= 0.17	St. Dev. = 0.01
Annual interaction nestedness (NODF)	= 10.54	St. Dev. = 4.38

✓ - Same or greater than reference mean.
↓ - Below reference mean, but within 1 St. Dev.
✗ - Greater than 1 St. Dev. below the mean.
👍 - Restoration successful (scored ≥ 16).
☞ - Restoration currently unsuccessful, but no remediation required (scored 8 <16).
👎 - Restoration unsuccessful. Remediation required (scored ≤ 8).

The criteria for determining whether restored landfill sites were successful at supporting specific groups of flower-visiting insects was whether more sites supported above the reference mean than below (given buffering for 1 St. Dev.). Any values greater than one standard deviation below the reference mean were deemed to be unsuccessfully restored. For groups of flower-visiting insects, the restored landfill sites can be deemed: successfully restored; currently unsuccessful, but not critical; or unsuccessful and remediation required.

Overall, taking the results from both 2007 and 2008 into consideration, those groups of flower-visiting insects for which restored landfill are successful are Bumblebees, Flies and Hoverflies (Tables 6.03 & 6.04). Those groups for which the restored landfill sites can be seen as unsuccessful in supporting are Butterflies and Beetles (Tables 6.03 & 6.04). As described previously in Chapter 4, reasons for the idiosyncratic response in different groups is likely attributed to habitat variables which were not recorded or relative dispersal and colonisation ability of the taxa.

Table 6.03 Summary of restoration success for species richness of flower visitor groups on restored landfill sites 2007 (values are compared to reference site means).

Restored landfill site	Bees	Beetles	Bumblebees	Butterflies	Flies	Hoverflies	Other
Bletchley	2 ✓	0 ↓	1 ×	0 ↓	3 ✓	2 ✓	1 ✓
Brixworth	1 ↓	0 ↓	3 ✓	0 ↓	1 ↓	7 ✓	5 ✓
Brogborough	1 ↓	0 ↓	3 ✓	0 ↓	1 ↓	4 ✓	1 ✓
Cranford	2 ✓	0 ↓	3 ✓	0 ↓	2 ✓	4 ✓	0 ↓
Harlestone	1 ↓	1 ✓	4 ✓	0 ↓	0 ×	1 ↓	0 ↓
Kettering	0 ×	1 ✓	4 ✓	0 ↓	3 ✓	2 ✓	1 ✓
Kilsby	0 ×	0 ↓	0 ×	0 ↓	0 ×	0 ×	1 ✓
Sidegate Lane	0 ×	0 ↓	0 ×	0 ↓	1 ↓	0 ×	0 ↓
Wootton	1 ↓	0 ↓	1 ×	0 ↓	3 ✓	3 ✓	0 ↓
Success for group	☞	♀	♂	♀	♂	♂	♂

Reference mean species richness for flower visitor insect groups per site: 2007

Bees	= 1.22 St. Dev. = 0.97
Beetles	= 0.11 St. Dev. = 0.33
Bumblebees	= 2.33 St. Dev. = 0.71
Butterflies	= 0.78 St. Dev. = 1.09
Flies	= 1.67 St. Dev. = 1.12
Hoverflies	= 1.89 St. Dev. = 0.93
Other	= 0.33 St. Dev. = 0.50

✓ - Same or greater than reference mean.
↓ - Below reference mean, but within 1 St. Dev.
× - Greater than 1 St. Dev. below the mean.
♂ - Restoration successful.
☞ - Restoration currently unsuccessful, but no remediation required.
♀ - Restoration unsuccessful. Remediation required.

Table 6.04 Summary of restoration success for species richness of flower visitor groups on restored landfill sites 2007 (values are compared to reference site means).

Restored landfill site	Bees	Beetles	Bumblebees	Butterflies	Flies	Hoverflies	Other
Brixworth	2 ✓	3 ✓	4 ✓	3 ✓	4 ✓	13 ✓	4 ✓
Sidegate Lane	0 ×	0 ×	1 ×	0 ×	0 ×	4 ×	1 ×
Wootton	3 ✓	1 ✓	4 ✓	2 ↓	2 ↓	11 ✓	1 ×
Success for group	👍	👍	👍	☞	☞	👍	👎

Reference mean species richness for flower visitor insect groups per site: 2007

Bees	= 1.33 St. Dev. = 0.58
Beetles	= 0.67 St. Dev. = 0.58
Bumblebees	= 3.67 St. Dev. = 1.53
Butterflies	= 2.33 St. Dev. = 2.31
Flies	= 3.00 St. Dev. = 3.46
Hoverflies	= 7.67 St. Dev. = 1.15
Other	= 2.33 St. Dev. = 0.58

✓ - Same or greater than reference mean.
↓ - Below reference mean, but within 1 St. Dev.
× - Greater than 1 St. Dev. below the mean.
👍 - Restoration successful.
☞ - Restoration currently unsuccessful, but no remediation required.
👎 - Restoration unsuccessful. Remediation required.

Conservation implications of restoring landfill sites for flower-visiting insects

This thesis has shown that restored landfill sites can support a relatively high diversity of plant and insect species. Although the total numbers of species differ, the trends in abundance and richness of plants and insects are similar in many ways and closely linked.

The potential of restored landfill sites is quite comparable to local nature reserves in terms of distribution and land area. There are approximately 2,200 landfill sites covering 28,000 ha. in England and Wales (Environment Agency, 2007a) and there are more than 1400 local nature reserves covering 35000 ha. in England (Natural England, 2009).

The even distribution throughout the landscape means that landfill sites have the potential for supporting great agricultural pollination service provision (Figure 6.01). When a 2km buffer is applied to landfill sites we see that the majority of the country is covered. With the effects of climate change altering the ranges of insect species, the restored sites may be utilised as stepping stones across the UK. The restoration of habitat sites may allow the movement of individual flower-visiting insects from one area to another, in so doing increasing or moving their habitat ranges. If restored sites provide the foraging and nesting needs of insects, networks of small reserves may hold important potential for sustaining considerable pollinator richness and the ecological services they provide (Cane, 2001). The evidence overall of the persistence of pollinating insects of relatively rich diversity and abundance on modest sized restored habitats promises a practical solution for the conservation of populations of flower-visiting insects.

Figure 6.01 Potential extent of pollination service provision when a 2 km buffer zone is applied to landfill sites in Great Britain (Elliott et al., 2001). Based on 9565 landfill sites operational at some time between 1982 and 1997.

Restored landfill sites can operate as long term wildlife habitats areas within the wider agricultural environment, providing areas free of agrochemical use. Restored landfill sites should be managed to maximise the floral resource provision in the autumn, when they may be absent within the landscape due to late summer grazing and mowing. If habitats are managed to provide a seasonal succession of suitable forage plants they can encourage bee and bumblebee populations (Fussell and Corbet, 1992; Corbet, 1995).

In the agricultural landscape, the diversity of flowering plants and pollinating insects are affected by a number of habitat attributes. For both insects and plants, it is important that these factors be integrated into their conservation measures. Community development and growth in a restored habitat requires a variety of plant mutualists. For plant reproduction, flower visitor richness and abundance must be adequate for the number and types of flowers, whether they have been artificially introduced or naturally set. Plant species that require specialised pollinator species may suffer the greatest limitation of seed set. Successful pollination will depend on the rates of invasion of the flower-visiting insects to the site. Protocols for restoration must include provisions for flower-visiting insects, as well as for the habitat requirements of the plant community. Giving due attention to the critical role of mutualists will increase the speed and likelihood of successful restoration, at no additional costs.

A landscape approach to restoration, may address issues related to practical constraints in restoration practice, such as ensuring that the establishment and recruitment of flora and fauna, is possible within the spatial configuration of the landscape. A study of agricultural field margins for example, found that small areas of habitat with flowering plants can be very effective at attracting and supporting pollinator populations (Lagerlof et al., 1992). Conservation and creation of bee habitats could be the best way of reversing the declines in pollinator populations. However, there is more to conserving an insect's habitat than food alone, and it possible that suitable nesting opportunities generally limit pollinating insect's abundance. Most flower visitors are generalists but have specific requirements for nesting. A study in the US has shown that bumblebee abundance in urban parks is limited by nest site availability (McFrederick and LeBuhn,

2006) and a recent study in the UK has shown that bumblebee nests appear to be more common in gardens than they are in the countryside (Osborne et al., 2008a). This may reflect a paucity of suitable nesting habitat in the rural environment. In this study, the correlation between flower-visiting insects and flowering plant richness and floral abundance does highlight that the floral resources are important, and that the presence of pollinator beneficial habitat features may be insufficient to significantly influence flower-visiting insect abundance or species richness. Encouraging pollinating insects to nest on restored landfill sites by offering suitable nesting habitat in combination with plentiful forage may help to ensure efficient pollination of wild and agricultural plants.

The maintenance of a healthy and diverse population of pollinating insects in the rural environment may ensure maximum yields of agricultural plants (Kremen et al., 2004), and may also be of great value for the survival and propagation of wildflowers (Osborne and Williams, 1996). Of recent development relating to this topic is the proposal of the world's first 'Pollination Park' on a restored landfill site in the city of Guelph, Ontario, Canada. Here Guelph's 40 ha. Eastview landfill site is being turned into a habitat for pollinators by designing the restoration to include plant species that attract pollinators (Pollination Guelph, 2009).

Recommendations for landfill site operators and the waste industry

Operators of landfill sites should consider their development in a holistic manner from the outset, including their working life, restoration and after use of the site; and this is now part of the government guidance (Watson and Hack, 2000). If operators are from the start proactive towards habitat restoration for wildlife then success is more likely and cost effective. Detailed planning from the outset may highlight valuable habitat on the sites, and targeting their protection may lead to speedier post-restoration colonisation. The restoration process may also be cheaper with local public engagement, potentially encouraging seed provision and management by local conservation groups. The advantages of wildlife habitat creation for landfill operators include its public relation value.

Restored landfill sites are relatively homogenous, having had graded soils and typically a seed mix over spread them. As such their best restoration may be to avoid the temptation to try and restore to a number of habitats and instead restore one large block of habitat. The restoration should aim to create a habitat suitable to the hydrology and topography and be appropriate to the surrounding landscape. Following is a number of recommendations aimed at restoration design and ongoing management practice on closed landfill sites, targeting habitat provision for flower-visiting pollinating insects.

Little is known about the life history of the majority of pollinating flower visitor species and so most advice here is based upon generalities. Most conservation advice is focused upon land management rather that restoration (e.g. Edwards, 1996). Consideration needs to be made whether to target the most effective pollinating insects, i.e. for the ecosystem service they provide to native and agricultural plants, or rarer species which are more threatened with extinction. Given the current wide-scale decline, restored habitats may be best at targeting their conservation efforts at the most general pollinating insects and letting NGO managed nature sites be concerned with rarer species with more specific habitat requirements. Focus of concern for restoration of habitats targeted for flower-visiting insects can therefore be on suitable general forage and nesting requirements. Restoration targeted at pollinating insects will not only benefit flower visitors but also numerous species supported by the plants. They will also potentially benefit plants in the surrounding landscape which benefit from their pollination services.

Soils

For many sites restoring in the general direction of species rich grassland may be beneficial. Soils of such sites typically have low concentrations of major plant nutrients particularly nitrogen and phosphorus. The low nutrient status, and relatively shallow soils will favour those plants which are drought tolerant species, typically deep rooted perennials (Hutchings and Stewart, 2002). Whilst selecting soils for overlaying, consideration should be made where possible to use soils which contain appropriate seed banks. This includes consideration of both storage and sources location. Use of lower fertility soils may benefit the floral assemblage through reducing competition from grasses. Compost should not be used for soil 'improvers' and bulk agents, and sand or gravel should be used to break up soils which are predominantly clay.

Seeding and plant selection

The typical cheap annual grass seed mixes used by landfill restoration may actually act as a 'nurse-crop'. They provide an immediate green cover which will help suppress weeds whilst the slower growing annuals establish (Hutchings et al., 2006). The assumption is that over time these initial species eventually will be excluded by the perennials, however the nurse crop will itself slow the establishment (Mitchley et al., 1996). Sowing may be undertaken via tractor mounted spreaders or hydro-seeding. With hydro-seeding, seed-water mulch is sprayed, which can effectively cover a large area quickly. Regular annual cutting on the restored landfill sites will prevent it growing too vigorously.

Where wildflower seed is used to create a flower-rich meadow, due care should be made towards selecting seeds which are locally sourced. Seeds may be collected from donor habitat sites using specialist machinery or from commercial suppliers. These can be used as a starter sward into which other species will gradually colonize. For specific targeted habitat for flower-visiting insects the inclusion of core generalist plants into the seed mix would be beneficial. Within this study the core plant species recommended for inclusion would be those less likely to occur naturally such as *Trifolium repens, Centaurea nigra, Lotus corniculatus, Ranunculus* spp., *Centaurea scabious.* Natural revegetation is likely to produce grassland habitat which is better suited to the site's physical and landscape conditions. Restoration may initially produce annual weeds, but with continued management, a stabilised rich floral community will develop, supporting numerous wild species.

Introducing plants via green hay, also called 'hay strewing', may be effective (Jones et al., 1995; Edwards et al., 2007; Kiehl and Pfadenhauer, 2007). Hay is collected from local donor sites after flowering, and contains seeds from many present plants. The hay is then spread soon after harvesting onto the new site. Local nature reserves may prove to be effective sources of material, and have freely available machinery and volunteers for hay cutting.

Planting surrounding hedgerows of native species such as hawthorn and blackthorn provide a habitat in their own right, and also increase bird dispersal of seed (Robinson and Handel, 1993). They can also provide an important nectar resource early in the season and nesting sites at their base (Kells and Goulson, 2003; Farkas and Zajácz, 2007). Where landfill restoration is continuing they can help reduce wind blown rubbish; ideally hedges which are tall and with thick bases should be encouraged.

Restored landfill habitats will develop in plant species richness and floral abundance overtime. Once floral resources are present above a certain threshold, and across each season, then focusing restoration efforts upon nesting habitat will achieve the most for flower-visiting insects (Kearns and Inouye, 1997).

Nesting habitats

Depending on taxa concerned, nesting sites may be subterranean, within dead wood and vegetation, or within the base of tussock grasses. Post-restoration creation of suitable subterranean excavations and earth faces may be problematic on restored landfill sites owing to the potential (albeit slight) risk of puncturing the geo-textile membrane. They could however be considered at the time of earth laying. There are usually numerous holes and cracks on site due to earth subsidence which some bee species may utilise (Figure 6.02). Restored landfill sites do not typically permit public access and so may be suitable for provision of nesting structures, without concern for vandalism.

Figure 6.02 Cracks and holes in the earth on restored landfill sites due to subsidence.

One of the simplest, cheapest and most effective measures for benefiting the conservation of flower-visiting insects could be the provision of Beetle Banks for nesting sites. Beetle Banks, defined as raised, tussock-grass sown strips, may be planted to encourage those bumblebees which utilize them, nesting at the bases (Lye et al., 2009). They are also used extensively by beetles and spiders for over wintering

(Marshall and Moonen, 1998). These 3-4m wide strips are typically planted with various grasses including Cock's-foot (*Dactylis glomerata*), Yorkshire fog (*Holcus lanatus*), or Red fescue (*Festuca rubra*), and are ideally left 2-3 years between mowing (Thomas, 2000).

Management

Management may include mowing or grazing. This is important for managing the fire risk on-site, and slowing vegetational succession. Mowing also helps suppress vigorously growing grasses which may out-compete the more diverse flowering perennials. As well as considering the proposed nesting habitats, important considerations relate to intensity and timing of mowing. Mowing is the most likely vegetation management to occur on restored landfill sites in order to reduce fire risks. Species richness on restored landfill sites significantly benefits from mowing without which succession will take the restored habitat into scrub and then woodland (Rebele and Lehmann, 2002).

Cutting should aim to leave floral resources when they are absent elsewhere within the environment and so a mid-summer cut is best, allowing time for re-flowering of plants before the autumn. A summer cut will also prevent the removal of larval food plants, eggs and caterpillars of butterfly species (Ellis et al., 2008). A great a height of cut as possible should be used, with the removal of clipping to reduce fertility ideally a few days after, to allow seed to fall and invertebrates to escape. On-site rotational cutting has been found to further promote insect diversity and abundance (Noordijk et al., 2009). Landfill site operators could consider collaborating with wildlife NGOs in the management of their sites, without giving access by public due to presence of potentially dangerous, fragile engineering structures. In exchange for the NGOs managing the sites, they would be allowed to develop the vegetation structure to better suit wildlife needs.

In summary for restoration practitioners and landfill site operators; landfill sites have great potential to be restored as wildlife habitats and specifically for flower-visiting insects. Relatively little required habitat and practice adjustment is required, particularly if this after-use is considered in the restoration planning. Poorer quality soils may be used, which are easier and cheaper to source. Ideally natural revegetation, should be allowed to occur, where not, then use of locally sourced soils or hay-strewing. For on-going management, mowing ideally should occur in late summer, preferably with rotation of bi-annual cut areas. Restored landfill sites have the potential to play a significant role in the conservation and achieve positive outcomes for both flower-visiting insects and the ecological services they provide.

Limitations of the data used in this thesis

The restored landfill sites have not been assessed fully with regards to their potential in the provision of specific habitat requirements for different groups, e.g. identified larval host plants, but have inferred potential of sites from insect presence.

Those flower-visiting insects recorded were not tested for whether they were residents or "tourists" of the sites. Flower-visiting insects may be primarily residing on-site, meeting all their requirements or may be resident off-site and utilising just food resources from within the sites.

The landfill sites were not a random selection of those within the UK, and hence making national implications from this research is difficult. However, nothing has been found to say that the landfill sites surveyed within this study are not representative of those across the UK.

This was a short term study, over two years of fieldwork, which may have been atypical years, in terms of climate or pollinator assemblage. The last two summers and the current one were wetter than normal(Met. Office, 2007; Met. Office, 2008; Met. Office, 2009). The current consensus is that we are suffering a pollinator decline at the moment (Biesmeijer et al., 2006; Grixti et al., 2009; Winfree et al., 2009). It is likely however, that both of these situations, atypical insect abundance and weather, would affect both types of site similarly and so comparison with reference sites will still allow examination of the restored landfill sites; their conservation potential for flower-visiting insects and determination of successful restoration.

Using paired sites reduced the need for spatial and landscape analysis of the surrounding landscape context, when comparing restored landfill sites to reference sites. However, for 'restored site-restored site' comparison nesting and forage value of surrounding landscape could have been measured.

Areas of further research to develop this study

Flowering plants

Research is needed following the ecological development of restored landfill sites over a longer time frame, from initial revegetation. Seed bank analysis is required of landfill site soils. Observations should be made to document successional change. Further investigation should be made into revegetation of restored landfill sites, comparing natural colonisation, seeding with imported seed mixes, and using locally sources seed mixes. Experimental plots using various alternative growth media e.g. recycled building waste, should be compared.

Flower-visiting insects

Research should be undertaken into an assessment of the pollinating services provided by flower-visiting insects supported on the restored landfill sites, particularly regarding agricultural crops. This could be assessed through bioassays plants over a gradient of distances from the landfill sites and measuring seed set, or pollen deposition.
Landfill sites and other restored habitats need to be assessed to determine the limiting factors on habitats for supporting flower-visiting insects on landfill sites.
Future studies should involve the creation of artificial assemblages of plants and pollinators, including behavioural studies involving development of additional plant species and observe changes in interaction structure and robustness.

Theoretical

The restoration framework needs further development to determine a fully compressive ecosystem services and interactions approach, both to determine successful restoration and highlight the importance of early consideration of species interactions and ecosystem, services.

Concluding statement

The findings from this research show that restored landfill sites can support as rich and abundant flower-visiting pollinating insect assemblage as reference wildlife sites. This is important both for their conservation and the ecosystem services they provide. There are over 2,000 restored landfill sites covering 28,000 ha. in the UK. This is a substantial resource of land and given their even distribution means that restored landfill sites can play an important role for the conservation of pollinating insects as habitats and as habitat-stepping stones. The artificial agricultural landscape developed over the last half-decade has been detrimental for our wildlife and in particular our flower-visiting insects. There is clear need to rectify this and strategically restoring wildlife habitats is paramount. These sites, if focused to provide floral and nesting resources can aid in the conservation effort of pollinating insects and the crucial ecosystem service they provide.

"The human race is challenged more than ever before to demonstrate our mastery, not over nature but of ourselves."
Rachel Carson (1907 – 1964), American nature writer.

Appendices

Appendix 1

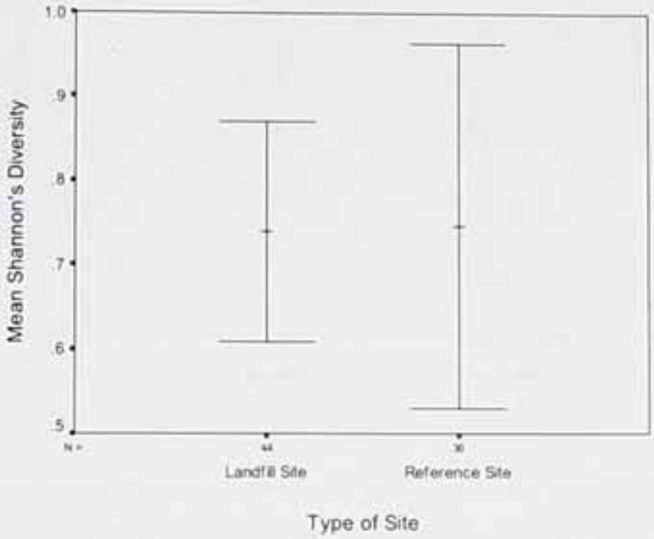
Table of flower sizes. All plants found on at least one of the study sites. sizes calculated from personal measurements and literature (Knuth, 1906-1909; Rose and O'Reilly, 2006).

Common name	Latin name	Flower Size (mm^2)
Yarrow	*Achillea millefolium*	15.71
Agrimony	*Agrimonia eupatoria*	20.42
Garlic Mustard	*Alliaria petiolata*	18.85
Cow Parsely	*Anthriscus sylvestris*	15.71
Hairy rock cress	*Arabis hirsuta*	11.00
Winter Cress	*Barbarea vulgaris*	25.13
Daisy	*Bellis perennis*	64.40
Hedge Bindweed	*Calystegia sepium*	172.79
Wavy Bitter cress	*Cardamine flexuosa*	9.42
Cuckoo-Flower	*Cardamine pratensis*	47.12
Common Knapweed	*Centaurea nigra*	94.25
Greater knapweed	*Centaurea scabiosa*	141.37
Common Centaury	*Centaurium erythraea*	34.56
Common Mouse-Ear	*Cerastium fontanum*	23.56
Chamomile	*Chamaemelum nobile*	67.54
Rosebay Willowherb	*Chamerion angustifolium*	78.54
Creeping Thistle	*Cirsium arvense*	31.42
Woolly Thistle	*Cirsium eriophorum*	109.96
Marsh Thistle	*Cirsium palustre*	39.27
Spear Thistle	*Cirsium vulgare*	102.10
Wild Basil	*Clinopodium vulgare*	47.12
Field Bindweed	*Convolvulus arvensis*	94.25
Marsh Hawk's-beard	*Crepis paludosa*	62.83
Beaked Hawks Beard	*Crepis vesicaria*	62.83
Common Spotted Orchid	*Dactylorhiza fuchsii*	36.00
Wild Carrot	*Daucus carota*	157.08
Wild Teasel	*Dipsacus fullonum*	157.08
Great Willowherb	*Epilobium hirsutum*	59.69
Meadowsweet	*Filipendula ulmaria*	21.99
GooseGrass	*Galium aparine*	6.28
Lady's Beadstraw	*Galium verum*	7.85
Cut-leaved Cranes Bill	*Geranium dissectum*	16.49
Ground ivy	*Glechoma hederacea*	54.98
Perforated St John's-wort	*Hypericium perforatum*	62.83
Field Scabious	*Knautia arvensis*	62.83
Prickly Lettuce	*Lactuca serriola*	37.70
White Dead Nettle	*Lamium album*	31.42
Red Dead Nettle	*Lamium purpureum*	31.42
Nipplewort	*Lapsana communis*	39.27
Grass Vetchling	*Lathyrus nissolia*	80.00
Meadow Vetchling	*Lathyrus pratensis*	51.84
Rough Hawkbit	*Leontodon hispidus*	102.10
Lesser Hawkbit	*Leontodon saxatilis*	70.69
Oxeye Daisy	*Leucanthemum vulgare*	133.52
Fairy Flax	*Linum catharticum*	15.71
Common Birds-foot Trefoil	*Lotus corniculatus*	25.13
Narrow Leaved Birds-foot Trefoil	*Lotus glaber*	21.99

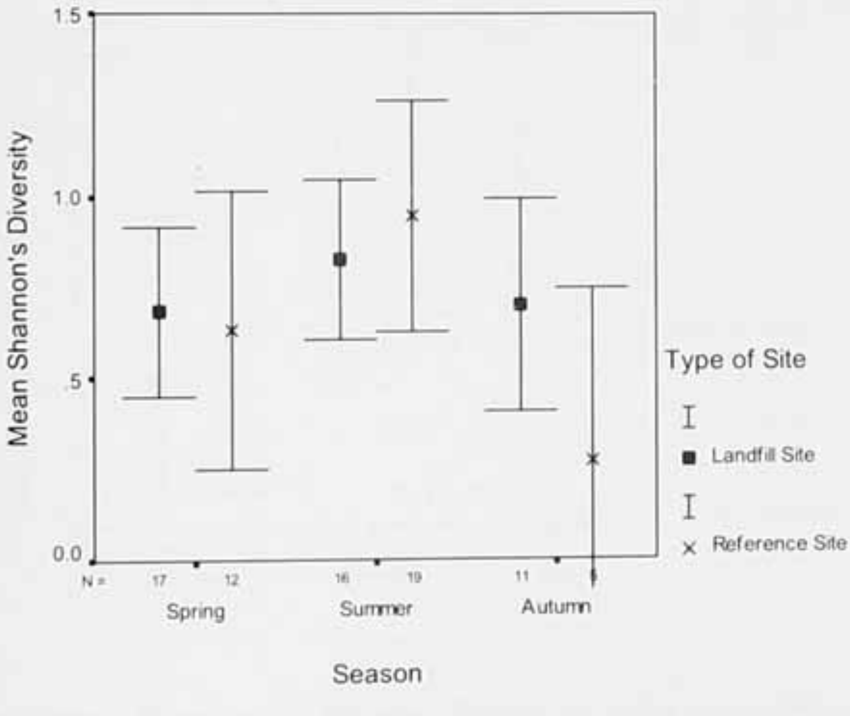
Black Medick	*Medicago lupulina*	15.71
Ribbed Melilot	*Melilotus officinalis*	17.28
Grape Hyacinth	*Muscari neglectum*	12.57
Field Forget-me-not	*Myosotis arvensis*	9.42
Early Forget-me-not	*Myosotis ramosissima*	9.42
Red Bartsia	*Odontites vernus*	28.27
Common Restharrow	*Ononis repens*	25.13
Spiny Restharrow	*Ononis spinosa*	18.85
Bee Orchid	*Ophrys apifera*	60.00
Green-winged Orchid	*Orchis morio*	75.00
Wild Parsnip	*Pastinaca sativa*	235.62
Redshank	*Persicaria masculosa*	15.71
Bristly Oxtongue	*Picris echioides*	78.54
Hawkweed Oxtongue	*Picris hieracioides*	78.54
Ribwort Plantain	*Plantago lanceolata*	12.57
Tormentil	*Potentilla erecta*	28.27
Creeping Cinquerfoil	*Potentilla reptans*	67.54
Cowslip	*Primula veris*	28.27
Selfheal	*Prunella vulgaris*	25.13
Meadow Buttercup	*Ranunculus acris*	47.12
Bulbous Buttercup	*Ranunculus bulbosus*	47.12
Lesser Celandine	*Ranunculus ficaria*	62.83
Creeping Buttercup	*Ranunculus repens*	47.12
Yellow Rattle	*Rhinanthus minor*	28.27
Bramble	*Rubus fruticosus*	31.42
Great Burnet	*Sangquisorba officinalis*	62.83
Hoary Ragwort	*Senecio erucifolius*	54.98
Common Ragwort	*Senecio jacobaea*	62.83
Field Madder	*Sherardia arvensis*	9.42
Bladder Campion	*Silene vulgaris*	61.26
Charlock	*Sinapis arvensis*	54.98
Smooth Sow Thistle	*Sonchus oleraceus*	70.69
Hedge Woundwort	*Stachy sylvatica*	15.71
Lesser Stitchwort	*Stellaria graminea*	37.70
Dandelion	*Taraxacum officinale*	125.66
Hop Trefoil	*Trifolium campestre*	39.27
Lesser Trefoil	*Trifolium dubium*	18.85
Red Clover	*Trifolium pratense*	62.83
White Clover	*Trifolium repens*	62.83
Wall Speed well	*Veronica arvensis*	14.14
Germander Speedwell	*Veronica chamaedrys*	15.71
Slender Speedwell	*Veronica filiformis*	25.13
Common Field Speedwell	*Veronica persica*	31.42
Thyme Leaved Speedwell	*Veronica serpyllifolia*	15.71
Tufted Vetch	*Vicia cracca*	25.13
Hairy Tare	*Vicia hirsuta*	15.71
Common Vetch	*Vicia sativa*	47.12
Bush Vetch	*Vicia sepium*	47.12
Smooth Tare	*Vicia tetrasperma*	12.57
Early Dog Violet	*Viola reichenbachiana*	56.55
Common Dog-violet	*Viola riviniana*	62.83

Appendix 2

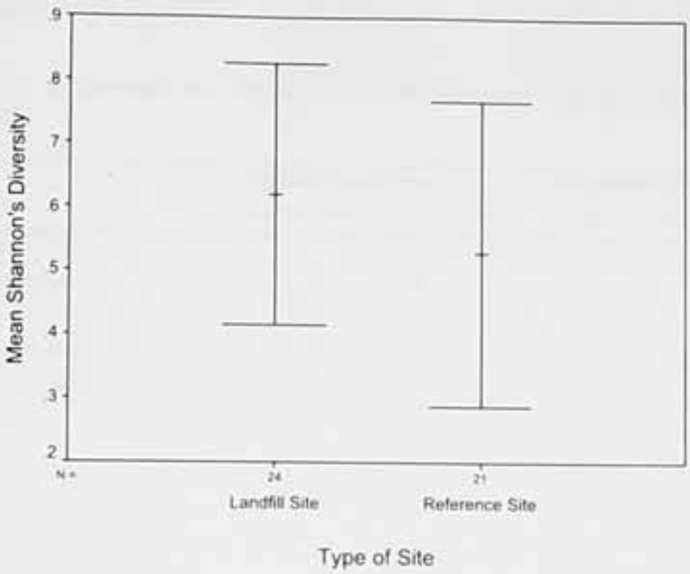
Shannon's diversity of flowering plant species



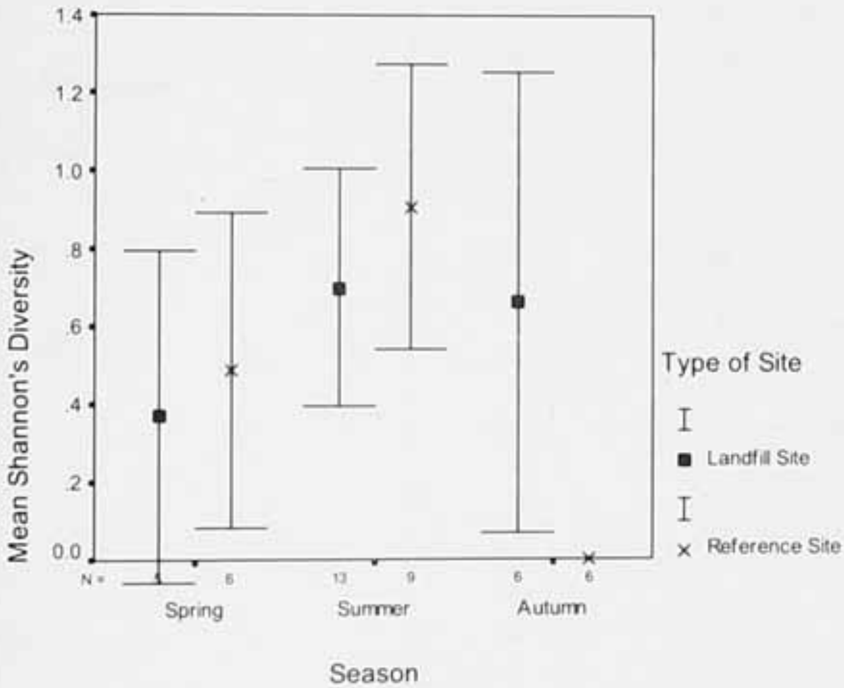
Mean flower Shannon's Diversity per transect for landfill sites and reference sites 2007 (±95% Confidence Limits). N= sample sizes. (one-way ANOVA ($F_{1,79}$ = 0.004 p= 0.951)



Mean seasonal flowering plant Shannon's Diversity per transect for landfill sites and reference sites 2007 (±95% Confidence Limits). N= sample sizes. (ANOVA; Landfill sites across seasons $F_{2,43}$=0.491 p=0.615, Reference sites across seasons $F_{2,35}$=2.718 p=0.081. Independent Samples T-Test (2-tailed) Spring Landfill vs. Reference t= 0.266 df=27 p=0.792, Summer Landfill vs. Reference t=-0.657 df=30.686 p=0.516, Autumn Landfill vs. Reference t=1.901 df=14 p=0.078.



Mean flower Shannon's Diversity per transect for landfill sites and reference sites 2008 (±95% Confidence Limits). N= sample sizes. (one-way ANOVA ($F_{1,44}$ = 0.373 p= 0.544)



Mean seasonal flowering plant Shannon's Diversity per transect for landfill sites and reference sites 2008 (±95% Confidence Limits). N= sample sizes. (ANOVA; Landfill sites across seasons $F_{2,23}$=0.834p=0.448, Reference sites across seasons $F_{2,20}$=10.463 p=0.001. Independent samples T-Test (2-tailed) Spring Landfill vs. Reference t= -0.529 df=9 p=0.610, Summer Landfill vs. Reference t=-0.975 df=20 p=0.341. Mann-Whitney 2-Tailed independent test: Autumn Landfill vs. Reference U=6.00 p=0.022.

Appendix 3

Table of presence and frequency of flower-visitor species on restored landfill and reference sites 2007

	Restored landfill sites		Reference sites	
	number of sites	present	number of sites	present
Honey Bee	6	1	5	1
Calliopum spp.	6	1	5	1
B terrestris	6	1	4	1
B lapidarius	5	1	8	1
B Pascorum	5	1	6	1
H7	5	1	3	1
H1	4	1	1	1
H8	3	1	3	1
O1	3	1	1	1
H5	3	1	0	0
H10	2	1	2	1
H9	2	1	2	1
F2	2	1	1	1
O7	2	1	1	1
B3	2	1	0	0
Psithryus sylvestris	2	1	0	0
H12	1	1	3	1
F5	1	1	2	1
Thymelicus sylvestris	1	1	1	1
H11	1	1	1	1
B pratorum	1	1	0	0
F10	1	1	0	0
F11	1	1	0	0
F7	1	1	0	0
H13	1	1	0	0
H3	1	1	0	0
H4	1	1	0	0
O2	1	1	0	0
O4	1	1	0	0
O6	1	1	0	0
B1	0	0	2	1
B4	0	0	2	1
F6	0	0	2	1
F1	0	0	2	1
BF2	0	0	2	1
B2	0	0	1	1
O5	0	0	1	1
O3	0	0	1	1
H6	0	0	1	1
H2	0	0	1	1
H14	0	0	1	1
F9	0	0	1	1
F4	0	0	1	1
F3	0	0	1	1
BF5	0	0	1	1

BF4	0	0	1	1
BF3	0	0	1	1
BF1	0	0	1	1
BB7	0	0	1	1
BB8	0	0	1	1

Appendix 4

Table of presence and frequency of flower-visitor species on restored landfill and reference sites 2008

	Restored landfill sites		Reference sites	
	number of sites	present	number of sites	present
Bombus lapidarius	3	1	3	1
Bombus pascuorum	3	1	3	1
Bombus pratorum	3	1	3	1
Bombus terrestris / lucorum	3	1	3	1
Ferdinandea ruficornis	3	1	3	1
Chaemaesyrphus scaevoides	3	1	2	1
Macrophya montana	3	1	2	1
Oedemera nobilis	3	1	1	1
Sphaerophoria scripta	3	1	1	1
Calliopum spp.	2	1	3	1
Syrphus vitripennis / ribesii	2	1	3	1
Pipizini sp.	2	1	2	1
Psilota anthracina	2	1	2	1
Chrysotoxum bicinctum	2	1	1	1
Episyrphus balteatus	2	1	0	0
Eriothrix rufomaculata	2	1	0	0
Eristalis cryptarum	2	1	0	0
Eristalis tenax	2	1	0	0
Macropis europaea	1	1	3	1
Maniola jurtina	1	1	3	1
Anthomyiidae sp.	1	1	2	1
Eristalis arbustorum	1	1	2	1
Helophilus pendulus	1	1	2	1
Megachile ssp.	1	1	2	1
Metasyrphus latifasciatus	1	1	2	1
Tenthredo notha	1	1	2	1
Eristalis intricarius	1	1	1	1
Muscina prolapsa	1	1	1	1
Pieris brassicae	1	1	1	1
Pyronia tithonus	1	1	1	1
Zygaena filipendulae	1	1	1	1
Aglais urticae	1	1	0	0
Apis mellifera	1	1	0	0
Chelostoma campanularum	1	1	0	0
Malachius bipustulatus	1	1	0	0
Osmia rufa	1	1	0	0
Osmia sp. *1*	1	1	0	0
Parhelophilus frutetorum	1	1	0	0

Polyommatus icarus	1	1	0	0
Rhagonycha fulva	1	1	0	0
Anasimyia lineata	0	0	1	1
Bombus hortorum	0	0	1	1
Calliphora vomitoria	0	0	1	1
Coenonympha pamphilus	0	0	1	1
Cryptocephalus hypochaeridis	0	0	1	1
Empis livida	0	0	1	1
Fly 1	0	0	1	1
Melanargia galathea	0	0	1	1
Meligethes aeneus	0	0	1	1
Ochlodes sylvanus	0	0	1	1
Panemeria tenebrata	0	0	1	1
Pollenia rudis	0	0	1	1
Sarcophaga carnaria	0	0	1	1

Appendix 5

Nestedness metrics values Temperature for Landfill and reference sites 2007.

('na' - matrix is too small for analysis.)

Year	Site	Type	T	Temperature Null models			
				T(Er)	P(Er)	T(Ce)	P(Ce)
2007	Barnes Meadow	Reference	44.07	41.99	0.57	37.16	0.69
2007	Bletchley	Landfill	53.28	45.37	0.70	40.10	0.82
2007	Blue Lagoon	Reference	12.13	37.73	0.03	27.94	0.21
2007	Brixworth	Landfill	43.00	46.73	0.35	43.42	0.48
2007	Brogborough	Landfill	22.56	45.00	0.02	36.86	0.14
2007	Cranford	Landfill	28.63	46.31	0.09	33.78	0.37
2007	Ditchford	Reference	42.68	37.18	0.66	30.96	0.72
2007	Draycote	Reference	32.19	34.90	0.44	30.74	0.57
2007	Glebe Meadow	Reference	38.99	45.55	0.27	39.78	0.48
2007	Harlestone	Landfill	28.01	39.61	0.22	31.18	0.44
2007	Kettering	Landfill	28.77	41.33	0.06	40.97	0.14
2007	Kilsby	Landfill	44.27	48.11	0.34	43.18	0.55
2007	Pitsford	Reference	41.68	47.13	0.35	40.78	0.55
2007	River Ise	Reference	45.32	43.31	0.55	38.39	0.67
2007	Scrubfield	Reference	55.04	44.79	0.81	40.70	0.83
2007	Sidegate	Landfill	79.36	43.11	0.99	39.44	0.99
2007	Twywell	Reference	44.07	41.99	0.57	37.16	0.69
2007	Wootton	Landfill	53.28	45.37	0.70	40.10	0.82

Appendix 6

Nestedness metrics values Temperature for Landfill and reference sites 2008.

('na' - matrix is too small for analysis.)

Season	Site	Type	T	Temperature Null models				Matrix
				(Er)	P(Er)	T(Ce)	P(Ce)	size
Spring	Barnes Meadow	Reference	42.97	49.18	0.30	41.04	0.57	78
Summer	Barnes Meadow	Reference	60.09	44.74	0.84	40.83	0.88	40
Autumn	Barnes Meadow	Reference	na	na	na	na	na	0
Annual	Barnes Meadow	Reference	40.75	51.94	0.13	42.09	0.48	140
Spring	Brixworth	Landfill	19.07	112.09	0.05	65.31	0.31	20
Summer	Brixworth	Landfill	15.23	38.56	0.00	34.26	0.00	312
Autumn	Brixworth	Landfill	18.63	42.33	0.02	33.22	0.14	45
Annual	Brixworth	Landfill	21.45	38.84	0.00	33.68	0.01	495
Spring	Ditchford	Reference	.	.	.	.	.	1
Summer	Ditchford	Reference	28.29	59.56	0.00	29.87	0.49	105
Autumn	Ditchford	Reference	.	.	.	.	.	1
Annual	Ditchford	Reference	28.36	57.34	0.00	31.46	0.44	132
Spring	Pitsford	Reference	18.89	32.20	0.23	25.81	0.41	21
Summer	Pitsford	Reference	41.41	45.88	0.32	41.62	0.48	128
Autumn	Pitsford	Reference	na	na	na	na	na	0
Annual	Pitsford	Reference	37.49	43.83	0.21	41.42	0.34	210
Spring	Sidegate lane	Landfill	0.00	0.00	1.00	0.00	1.00	2
Summer	Sidegate lane	Landfill	20.67	39.31	0.51	22.14	0.72	8
Autumn	Sidegate lane	Landfill	na	na	na	na	na	0
Annual	Sidegate lane	Landfill	67.39	39.40	0.96	35.02	0.90	18
Spring	Wootton	Landfill	18.26	27.72	0.36	22.80	0.53	12
Summer	Wootton	Landfill	33.59	47.29	0.16	36.01	0.45	55
Autumn	Wootton	Landfill	33.54	48.05	0.12	36.87	0.41	65
Annual	Wootton	Landfill	37.26	51.93	0.04	44.07	0.23	192

Appendix 7

Species identities for Flower-insect visitor interaction structure in 2008 on Brixworth restored landfill and Pitsford reference site

Plant 1	*Achillea millefolium*
Plant 2	*Bellis perennis*
Plant 3	*Cardamine pratensis*
Plant 4	*Cirsium arvense*
Plant 5	*Cirsium vulgare*
Plant 6	*Heracleum* sp.
Plant 7	*Leontodon hispidus*
Plant 8	*Leucanthemum vulgare*
Plant 9	*Lotus corniculatus*
Plant 10	*Picris echioides*
Plant 11	*Picris hieracioides*
Plant 12	*Potentilla reptans*
Plant 13	*Ranunculus repens*
Plant 14	*Senecio jacobaea*
Plant 15	*Taraxacum officinale*
Plant 16	*Cirsium palustre*

Plant 17	*Hypochaeris radicata*
Plant 18	*Leontodon saxatilis*
Plant 19	*Ranunculus acris*
Plant 20	*Ranunculus bulbosus*
Plant 21	*Rhinanthus minor*
Plant 22	*Trifolium pratense*
Insect 1	*Episyrphus balteatus*
Insect 2	*Sphaerophoria scripta*
Insect 3	*Bombus lapidarius*
Insect 4	*Calliopum spp*
Insect 5	*Eristalis tenax*
Insect 6	*Chrysotoxum bicinctum*
Insect 7	*Ferdinandea ruficornis*
Insect 8	*Helophilus pendulus*
Insect 9	*Aglais urticae*
Insect 10	*Bombus terrestris / lucorum*
Insect 11	*Eristalis cryptarum*
Insect 12	*Oedemera nobilis*
Insect 13	*Pipizini* sp.
Insect 14	*Syrphus vitripennis / ribesii*
Insect 15	*Anthomyiidae* sp.
Insect 16	*Bombus pascuorum*
Insect 17	*Bombus pratorum*
Insect 18	*Calliphora vomitoria*
Insect 19	*Chaemaesyrphus scaevoides*
Insect 20	*Chelostoma campanularum*
Insect 21	*Eristalis intricarius*
Insect 22	*Macrophya montana*
Insect 23	*Malachius bipustulatus*
Insect 24	*Megachile* ssp.
Insect 25	*Metasyrphus latifasciatus*
Insect 26	*Muscina prolapsa*
Insect 27	*Osmia* sp. 1
Insect 28	*Psilota anthracina*
Insect 29	*Pyronia tithonus*
Insect 30	*Rhagonycha fulva*
Insect 31	*Zygaena filipendulae*
Insect 32	*Eristalis arbustorum*
Insect 33	*Macropis europaea*
Insect 34	*Maniola jurtina*
Insect 35	*Meligethes aeneus*
Insect 36	*Panemeria tenebrata*
Insect 37	*Pieris brassicae*
Insect 38	*Tenthredo notha*

Appendix 8

Species identities for Flower-insect visitor interaction structure in 2008 on Wootton restored landfill and Barnes Meadow reference site

Plant 1	*Anthriscus sylvestris*
Plant 2	*Cerastium fontanum*
Plant 3	*Cirsium palustre*
Plant 4	*Picris echioides*
Plant 5	*Picris hieracioides*
Plant 6	*Ranunculus acris*
Plant 7	*Ranunculus bulbosus*
Plant 8	*Ranunculus repens*
Plant 9	*Senecio jacobaea*
Plant 10	*Taraxacum officinale*
Plant 11	*Trifolium pratense*
Plant 12	*Trifolium repens*
Insect 1	*Anasimyia lineata*
Insect 2	*Apis mellifera*
Insect 3	*Bombus hortorum*
Insect 4	*Bombus lapidarius*
Insect 5	*Bombus pascuorum*
Insect 6	*Bombus pratorum*
Insect 7	*Bombus terrestris / lucorum*
Insect 8	*Chaemaesyrphus scaevoides*
Insect 9	*Episyrphus balteatus*
Insect 10	*Eristalis intricarius*
Insect 11	*Eriothrix rufomaculata*
Insect 12	*Eristalis arbustorum*
Insect 13	*Eristalis cryptarum*
Insect 14	*Eristalis tenax*
Insect 15	*Ferdinandea ruficornis*
Insect 16	*Calliopum spp*
Insect 17	*Macrophya montana*
Insect 18	*Macropis europaea*
Insect 19	*Metasyrphus latifasciatus*
Insect 20	*Maniola jurtina*
Insect 21	*Ochlodes sylvanus*
Insect 22	*Oedemera nobilis*
Insect 23	*Osmia rufa*
Insect 24	*Parhelophilus frutetorum*
Insect 25	*Pieris brassicae*
Insect 26	*Pipizini* sp.
Insect 27	*Psilota anthracina*
Insect 28	*Sphaerophoria scripta*
Insect 29	*Syrphus vitripennis / ribesii*
Insect 30	*Tenthredo notha*

Appendix 9

Species identities for Flower-insect visitor interaction structure in 2008 on Sidegate Lane restored landfill and Ditchford reference site

Plant 1	*Cardamine pratensis*
Plant 2	*Ranunculus acris*
Plant 3	*Ranunculus repens*
Plant 4	*Sangquisorba officinallis*
Plant 5	*Taraxacum officinale*
Plant 6	*Trifolium pratense*
Plant 7	*Trifolium repens*
Plant 8	*Anthomyiidae* sp.
Insect 1	*Bombus pascuorum*
Insect 2	*Bombus terrestris / lucorum*
Insect 3	*Chaemaesyrphus scaevoides*
Insect 4	*Chrysotoxum bicinctum*
Insect 5	*Empis livida*
Insect 6	*Eristalis arbustorum*
Insect 7	*Ferdinandea ruficornis*
Insect 8	Fly 1
Insect 9	*Calliopum spp*
Insect 10	*Helophilus pendulus*
Insect 11	*Macrophya montana*
Insect 12	*Macrophya montana*
Insect 13	*Macropis europaea*
Insect 14	*Maniola jurtina*
Insect 15	*Megachile ssp.*
Insect 16	*Metasyrphus latifasciatus*
Insect 17	*Muscina prolapsa*
Insect 18	*Pipizini* sp.
Insect 19	*Pollenia rudis*
Insect 20	*Psilota anthracina*
Insect 21	*Sarcophaga carnaria*
Insect 22	*Sphaerophoria scripta*
Insect 23	*Syrphus vitripennis / ribesii*
Insect 24	*Tenthredo notha*

References

References

"If you have knowledge, let others light their candles at it"

Margaret Fuller (1810 – 1850)

"About the most originality that any writer can hope to achieve honestly is to steal with good judgment."

Josh Billings (1818 – 1885)

AIZEN, M. A. & FEINSINGER, P. (1994) Forest fragmentation, pollination, and plant reproduction in a chaco dry forest, Argentina. *Ecology,* **75,** 330-351.

ALARCÓN, R., WASER, N. M. & OLLERTON, J. (2008) Year-to-year variation in the topology of a plant-pollinator interaction network. *Oikos,* **117,** 1796-1807.

ALLEE, W. C., EMERSON, A. E., PARK, O., PARK, T. & SCHMIDT, K. P. (1949) *Principles of animal ecology,* PA, Philadelphia.

ALLEN-WARDELL, G., BERNHARDT, P., BITNER, R., BURQUEZ, A., BUCHMANN, S., CANE, J., COX, P. A., DALTON, V., FEINSINGER, P., INGRAM, M., INOUYE, D., JONES, C. E., KENNEDY, K., KEVAN, P., KOOPOWITZ, H., MEDELLIN, R., MEDELLIN-MORALES, S., NABHAN, G. P., PAVLIK, B., TEPEDINO, V., TORCHIO, P. & WALKER, S. (1998) The potential consequences of pollinator declines on the conservation of biodiversity and stability of food crop yields. *Conservation Biology,* **12,** 8-17.

ALMEIDA-NETO, M., GUIMARAES, P., GUIMARAES, P. R., LOYOLA, R. D. & ULRICH, W. (2008) A consistent metric for nestedness analysis in ecological systems: reconciling concept and measurement. *Oikos,* **117,** 1227 - 1239.

ALMEIDA-NETO, M., GUIMARAES, P. R. & LEWINSOHN, T. M. (2007) On nestedness analyses: rethinking matrix temperature and anti-nestedness. *Oikos,* **116,** 716-722.

ANDERSON, J. M. & INGRAM, J. S. I. (1993) *Tropical soil biology and fertility. A handbook of methods,* CAB International, Wallingford, UK.

ANON. (1994) *The UK Biodiversity Action Plan,* HMSO, London.

ATHY, E. R., KEIFFER, C. H. & STEVENS, M. H. (2006) Effects of mulch on seedlings and soil on a closed landfill. *Restoration Ecology,* **14,** 233-241.

ATMAR, W. & PATTERSON, B. D. (1995) The nestedness calculator: a visual basic program, including 294 presence-absence matrices. Chicago, AICS Research, Inc. University Park, NM and The Field Museum.

BACKMAN, J. P. C. & TIAINEN, J. (2002) Habitat quality of field margins in a Finnish farmland area for bumblebees (*Hymenoptera: Bombus* and *Psithyrus*). *Agriculture, Ecosystems & Environment,* **89,** 53-68.

BAER, S. G., COLLINS, S. L., BLAIR, J. M., KNAPP, A. K. & FIEDLER, A. K. (2005) Soil heterogeneity effects on tall-grass prairie community heterogeneity: An application of ecological theory to restoration ecology. *Restoration Ecology,* **13,** 413-424.

BANASZAK, J. (1980) Studies on methods of censusing the numbers of bees. *Polish Ecological Studies,* **6,** 355 - 366.

BASCOMPTE, J. & JORDANO, P. (2007) Plant-animal mutualistic networks: The architecture of biodiversity. *Annual Review of Ecology, Evolution, and Systematics,* **38,** 567-593.

BASCOMPTE, J., JORDANO, P., MELIAN, C. J. & OLESEN, J. M. (2003) The nested assembly of plant-animal mutualistic networks. *Proceedings of the National Academy of Sciences,* **100,** 9383-9387.

BASRI, H. (1998) An expert system for planning landfill restoration. *Water Science and Technology,* **37,** 211-217.

BATAGELJ, V. & MRVAR, A. (1998) Pajek-program for large network analysis. *Connections,* **21,** 47-57.

BAWA, K. S. (1990) Plant-pollinator interactions in tropical rain forests. *Annual Review of Ecology and Systematics,* **21,** 399-422.

BAYFIELD, N. G. (1995) Species selection and management for slope revegeation projects. In: *Vegetation and Slopes Stabilisation Protection and Ecology.* (Eds.) BARKER, D. H. Institution of Civil Engineers

BEIER, P. & NOSS, R. F. (1998) Do habitat corridors provide connectivity? *Conservation Biology,* **12,** 1241-1252.

BELL, S. S., FONSECA, M. S. & MOTTEN, L. B. (1997) Linking restoration and landscape ecology. *Restoration Ecology,* **5,** 318-323.

BENTON, T. G., VICKERY, J. A. & WILSON, J. D. (2003) Farmland biodiversity: is habitat heterogeneity the key? *Trends in Ecology & Evolution,* **18,** 182-188.

BIESMEIJER, J. C., ROBERTS, S. P. M., REEMER, M., OHLEMULLER, R., EDWARDS, M., PEETERS, T., SCHAFFERS, A. P., POTTS, S. G., KLEUKERS, R., THOMAS, C. D., SETTELE, J. & KUNIN, W. E. (2006) Parallel declines in pollinators and insect-pollinated plants in Britain and the Netherlands. *Science,* **313,** 351-354.

BOND, W. J. (1994) Do mutualisms matter? Assessing the impact of pollinator and disperser disruption on plant extinction. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences,* **344,** 83-90.

BOSCH, J., MARTÍN GONZÁLEZ, A. M., RODRIGO, A. & NAVARRO, D. (2009) Plant-pollinator networks: adding the pollinator's perspective. *Ecology Letters,* **12,** 409-419.

BROSI, B. J., ARMSWORTH, P. R. & DAILY, G. C. (2008) Optimal design of agricultural landscapes for pollination services. *Conservation Letters,* **1,** 27-36.

BROWN, V. K. & SOUTHWOOD, T. R. E. (1987) Secondary succession: patterns and strategies. In: *Colonization, Succession and Stability.* (Eds.) GRAY, A. J., CRAWLEY, M. J. & EDWARDS, D. J. Blackwell Scientific Publications, Oxford

BUCHMANN, S. L. & NABHAN, G. P. (1996) *The Forgotten Pollinators,* Island Press.

BURD, M. (1994) Bateman's principle and plant reproduction: the role of pollen limitation in fruit and seed set. *Botanical Review,* **60,** 83-139.

BURGOS, E., CEVA, H., PERAZZO, R. P. J., DEVOTO, M., MEDAN, D., ZIMMERMANN, M. & MARÍA DELBUE, A. (2007) Why nestedness in mutualistic networks? *Journal of Theoretical Biology,* **249,** 307-313.

CANE, J. H. (2001) Habitat fragmentation and native bees: a premature verdict? *Conservation Ecology,* **5,** 1-3.

CANT, E. T., SMITH, A. D., REYNOLDS, D. R. & OSBORNE, J. L. (2005) Tracking butterfly flight paths across the landscape with harmonic radar. *Proceedings of the Royal Society B-Biological Sciences,* **272,** 785-790.

CARTER, A. (1892) Evolution in methods of pollination *Botanical Gazette,* **17,** 72-78.

CARVELL, C. (2002) Habitat use and conservation of bumblebees (*Bombus* spp.) under different grassland management regimes. *Biological Conservation,* **103,** 33-49.

CARVELL, C., MEEK, W. R., PYWELL, R. F. & NOWAKOWSKI, M. (2004) The response of foraging bumblebees to successional change in newly created arable field margins. *Biological Conservation,* **118,** 327-339.

CARVELL, C., ROY, D. B., SMART, S. M., PYWELL, R. F., PRESTON, C. D. & GOULSON, D. (2006) Declines in forage availability for bumblebees at a national scale. *Biological Conservation,* **132,** 481-489.

CHAN, Y. S. G., CHU, L. M. & WONG, M. H. (1997) Influence of landfill factors on plants and soil fauna: An ecological perspective. *Environmental Pollution,* **97,** 39-44.

CHEPTOU, P. O. & AVENDANO, V. L. G. (2006) Pollination processes and the Allee effect in highly fragmented populations: consequences for the mating system in urban environments. *New Phytologist,* **172,** 774-783.

CHINERY, M. (2005) *Complete British insects,* Collins, London.

CLOUGH, Y., KRUESS, A. & TSCHARNTKE, T. (2007) Local and landscape factors in differently managed arable fields affect the insect herbivore community of a non-crop plant species. *Journal of Applied Ecology,* **44,** 22-28.

COLLA, S. & PACKER, L. (2008) Evidence for decline in eastern North American bumblebees (Hymenoptera: Apidae), with special focus on *Bombus affinis* Cresson. *Biodiversity and Conservation,* **17,** 1379-1391.

COLYER, C. N. & HAMMOND, C. O. (1951) *Flies of the British Isles,* Fredrick Warne & Co. Ltd. , London & New York.

CONNELL, J. H. (1978) Diversity in tropical rain forests and coral reefs. *Science,* **199,** 1302-1310.

CORBET, S. A. (1978) Bee visits and the nectar of *Echium vulgare* L. and *Sinapis alba* L. *Ecological Entomology,* **3,** 25-37.

CORBET, S. A. (1995) Insects, plants and succession: advantages of long-term set-aside. *Agriculture, Ecosystems & Environment,* **53,** 201-217.

CORBET, S. A. (2000) Conserving compartments in pollination webs. *Conservation Biology,* **14,** 1229-1231.

CORBET, S. A., UNWIN, D. M. & PRYS-JONES, O. E. (1979) Humidity, nectar and insect visits to flowers, with special reference to Crataegus, Tilia and Echium. *Ecological Entomology,* **4,** 9-22.

CORTINA, J., MAESTRE, F. T., VALLEJO, R., BAEZA, M. J., VALDECANTOS, A. & PEREZ-DEVESA, M. (2006) Ecosystem structure, function, and restoration success: Are they related? *Journal for Nature Conservation,* **14,** 152-160.

COUNTRYSIDE COMMISSION (1986) *Monitoring landscape change*, Cheltenham.

COURCHAMP, F., ANGULO, E., RIVALAN, P., HALL, R. J., SIGNORET, L., BULL, L. & MEINARD, Y. (2006) Rarity value and species extinction: The anthropogenic allee effect. *PLoS Biol* **4,** 2405 - 2410.

CROOK, C. S. (1992) The feasibility of tree planting on landfill containment sites. *Arboricultural Journal,* **16,** 229-241.

CRUDEN, R. W., MCCLAIN, A. M. & SHRIVASTAVA, G. P. (1996) Pollination biology and breeding system of *Alliaria petiolata* (Brassicaceae). *Bulletin of the Torrey Botanical Club,* **123,** 273-280.

DAFNI, A., KEVAN, P. G. & HUSBAND, B. C. (2005) *Practical pollination biology,* Environquest.

DAVIS, B. N. K. & COPPEARD, R. P. (1987) Restoration of a landfill site for amenity and wildlife: Soil conditions and grassland establishment. *Journal of the Science of Food and Agriculture,* **40,** 225-226.

DAVIS, B. N. K. & COPPEARD, R. P. (1989) Soil conditions and grassland establishment or amenity and wildlife on a restored landfill site. In: *Biological Habitat Reconstruction.* (Eds.) BUCKLEY, G. P. Belhaven Press, London.

DENSLOW, J. S. (1980) Patterns of plant species diversity during succession under different disturbance regimes. *Oecologia,* **46,** 18-21.

DIAMOND, J. M. (1970) Ecological consequences of island colonization by Southwest Pacific birds, I. Types of niche shifts. *Proceedings of the National Academy of Sciences of the United States of America,* **67,** 529-536.

DICKS, L. V., CORBET, S. A. & PYWELL, R. F. (2002) Compartmentalization in plant-insect flower visitor webs. *Journal of Animal Ecology,* **71,** 32-43.

DIXON, K. W. (2009) Pollination and Restoration. *Science,* **325,** 571-573.

DOBSON, A. P., BRADSHAW, A. D. & BAKER, A. J. M. (1997) Hopes for the future: Restoration ecology and conservation biology. *Science,* **277,** 515-522.

DONATH, T. W., HÖLZEL, N. & OTTE, A. (2003) The impact of site conditions and seed dispersal on restoration success in alluvial meadows. *Applied Vegetation Science,* **6,** 13-22.

DORMANN, C. F., SCHWEIGER, O., AUGENSTEIN, I., BAILEY, D., BILLETER, R., DE BLUST, G., DEFILIPPI, R., FRENZEL, M., HENDRICKX, F., HERZOG, F., KLOTZ, S., LIIRA, J., MAELFAIT, J.-P., SCHMIDT, T., SPEELMANS, M., VAN WINGERDEN, W. K. R. E. & ZOBEL, M. (2007) Effects of landscape structure and land-use intensity on similarity of plant and animal communities. *Global Ecology and Biogeography*, 1-14.

DRAMSTAD, W. (1996) Do bumblebees (Hymenoptera: Apidae) really forage close to their nests? *Journal of Insect Behavior,* **9,** 163-182.

DRYDEN, R. (1997) Habitat restoration project: Fact sheets and bibliographies. *English Nature*.

DUNNE, J. A., WILLIAMS, R. J. & MARTINEZ, N. D. (2002) Network structure and biodiversity loss in food webs: robustness increases with connectance. *Ecology Letters,* **5,** 558-567.

DUPONT, Y. L., HANSEN, D. M. & OLESEN, J. M. (2003) Structure of a plant-flower-visitor network in the high-altitude sub-alpine desert of Tenerife, Canary Islands. *Ecography,* **26,** 301-310.

DUPONT, Y. L., PADRÓN, B., OLESEN, J. M. & PETANIDOU, T. (2009) Spatio-temporal variation in the structure of pollination networks. *Oikos,* **118,** 1261-1269.

EARLE, D. F. & MCGOWAN, A. A. (1979) Evaluation and calibration of an automated rising plate meter for estimating dry matter yield of pasture. *Australian Journal of Experimental Agriculture,* **19,** 337-343.

EBELING, A., KLEIN, A.-M., SCHUMACHER, J., WEISSER, W. W. & TSCHARNTKE, T. (2008) How does plant richness affect pollinator richness and temporal stability of flower visits? *Oikos,* **117,** 1808-1815.

EDWARDS, A. R., MORTIMER, S. R., LAWSON, C. S., WESTBURY, D. B., HARRIS, S. J., WOODCOCK, B. A. & BROWN, V. K. (2007) Hay strewing, brush harvesting of seed and soil disturbance as tools for the enhancement of botanical diversity in grasslands. *Biological Conservation,* **134,** 372-382.

EDWARDS, M. (1996) Optimizing habitats for bees in the United Kingdom - a review of recent conservation action. In: *The conservation of bees.* (Eds.) MATHESON, A., BUCHMANN, S. L., O'TOOLE, C., WESTRICH, P. & WILLIAMS, I. H. Academic Press, London.

EHRENFELD, J. G. & TOTH, L. A. (1997) Restoration ecology and the ecosystem perspective. *Restoration Ecology,* **5,** 307-317.

ELLIOTT, P., BRIGGS, D., MORRIS, S., DE HOOGH, C., HURT, C., JENSEN, T. K., MAITLAND, I., RICHARDSON, S., WAKEFIELD, J. & JARUP, L. (2001) Risk of adverse birth outcomes in populations living near landfill sites. *British Medical Journal,* **323,** 363-368.

ELLIS, S., FOX, R. & WARREN, M. (2008) *Guidelines for managing urban habitats for butterflies,* Butterfly Conservation, Wareham, Dorset.

ENVIRONMENT AGENCY (2004) Technical guidance on capping and restoration of landfills.

ENVIRONMENT AGENCY (2006) What are landfill sites? Accessed; October 2006. From: http://www.environment-agency.gov.uk/yourenv/eff/1190084/resources_waste

ENVIRONMENT AGENCY (2007a) Capacity at permitted landfill sites in England and Wales: 2005 Accessed; October 2007. From: http://www.environment-agency.gov.uk/subjects/waste/1031954/315439/1720716/1746994/?version=1&lang=_e

ENVIRONMENT AGENCY (2007b) East of England waste sites Accessed; October 2007. From: http://www.environment-agency.gov.uk/subjects/waste/1031954/315439/1720716/1747765/1757474/?lang=_e

ENVIRONMENT AGENCY (2007c) Waste inputs to different management options in 2005 Accessed; October 2007. From: http://www.environment-agency.gov.uk/commondata/103196/1753133?referrer=/subjects/waste/1031954/315439/1720716/

ERZINCLIOGLU, Z. (1996) *Blowflies,* Richmond Publishing Co. Ltd. , Slough.

ETTALA, M. O. (1988) Short-rotation tree plantations at sanitary landfills. *Waste Management & Research,* **6,** 291-302.

ETTALA, M. O., YRJONEN, K. M. & ROSSI, E. J. (1988) Vegetation coverage at sanitary landfills in Finland. *Waste Management Research,* **6,** 281-289.

FALCY, M. R. & ESTADES, C. F. (2007) Effectiveness of corridors relative to enlargement of habitat patches. *Conservation Biology,* **21,** 1341-1346.

FALK, D. A., PALMER, M. A. & ZEDLER, J. B. (Eds.) (2006) *Foundations of restoration ecology,* Island Press, Washington.

FARKAS, Á. & ZAJÁCZ, E. (2007) Nectar production for the Hungarian honey industry. *European Journal of Plant Science and Biotechnology,* **1,** 125-151.

FARQUHAR, G. J. & ROVERS, F. A. (1973) Gas production during refuse decomposition. *Water, Air, & Soil Pollution,* **2,** 483-495.

FENNER, M. & THOMPSON, K. (2005) *The ecology of seeds,* Cambridge University Press.

FENSTER, C. B., ARMBRUSTER, W. S., WILSON, P., DUDASH, M. R. & THOMSON, J. D. (2004) Pollination syndromes and floral specialization. *Annual Review of Ecology, Evolution, and Systematics,* **35,** 375-403.

FISCHER, J. & LINDENMAYER, D. B. (2002) Treating the nestedness temperature calculator as a "black box" can lead to false conclusions. *Oikos,* **99,** 193-199.

FITTER, A. H. (1982) Influence of soil heterogeneity on the coexistence of grassland species. *The Journal of Ecology,* **70,** 139-148.

FITTER, A. H., FITTER, R. S. R., HARRIS, I. T. B. & WILLIAMSON, M. H. (1995) Relationships between first flowering date and temperature in the flora of a locality in central England. *Functional Ecology,* **9,** 55-60.

FITZPATRICK, Ú., MURRAY, T. E., PAXTON, R. J., BREEN, J., COTTON, D., SANTORUM, V. & BROWN, M. J. F. (2007) Rarity and decline in bumblebees - A test of causes and correlates in the Irish fauna. *Biological Conservation,* **136,** 185-194.

FONTAINE, C., COLLIN, C. L. & DAJOZ, I. (2008) Generalist foraging of pollinators: diet expansion at high density. *Journal of Ecology,* **96,** 1002-1010.

FONTAINE, C., DAJOZ, I., MERIGUET, J. & LOREAU, M. (2006a) Functional diversity of plant-pollinator interaction webs enhances the persistence of plant communities. *PLoS Biology,* **4,** e1.

FONTAINE, C., MERIGUET, J., LOREAU, M. & DAJOZ, I. (2006b) Diversity of plant-pollinator interaction and stability of ecosystems *MS -PARIS-* **22,** pp. 817-819.

FORTUNA, M. A. & BASCOMPTE, J. (2006) Habitat loss and the structure of plant-animal mutualistic networks. *Ecology Letters,* **9,** 281-286.

FORUP, M. L., HENSON, K. S. E., CRAZE, P. G. & MEMMOTT, J. (2008) The restoration of ecological interactions: plant-pollinator networks on ancient and restored heathlands. *Journal of Applied Ecology,* **45,** 742-752.

FORUP, M. L. & MEMMOTT, J. (2005a) The relationship between the abundances of bumblebees and honeybees in a native habitat. *Ecological Entomology,* **30,** 47-57.

FORUP, M. L. & MEMMOTT, J. (2005b) The restoration of plant-pollinator interactions in hay meadows. *Restoration Ecology,* **13,** 265-274.

FRANZÉN, M. & NILSSON, S. G. (2008) How can we preserve and restore species richness of pollinating insects on agricultural land? *Ecography,* **31,** 698 - 708.

FREEMARK, K. & BOUTIN, C. (1995) Impacts of agricultural herbicide use on terrestrial wildlife in temperate landscapes: A review with special reference to North America. *Agriculture, Ecosystems & Environment,* **52,** 67-91.

FUSSELL, M. & CORBET, S. A. (1992) Flower usage by bumble-bees: A basis for forage plant management. *The Journal of Applied Ecology,* **29,** 451-465.

GALLAI, N., SALLES, J.-M., SETTELE, J. & VAISSIÈRE, B. E. (2009) Economic valuation of the vulnerability of world agriculture confronted with pollinator decline. *Ecological Economics,* **68,** 810-821.

GARDENER, M. C. & GILLMAN, M. P. (2002) The taste of nectar-a neglected area of pollination ecology. *Oikos,* **98,** 552-557.

GATHMANN, A. & TSCHARNTKE, T. (2002) Foraging ranges of solitary bees. *The Journal of Animal Ecology,* **71,** 757-764.

GHAZOUL, J. (2005) Buzziness as usual? Questioning the global pollination crisis. *Trends in Ecology & Evolution,* **20,** 367-373.

GHAZOUL, J. (2006) Floral diversity and the facilitation of pollination. *Journal of Ecology,* **94,** 295-304.

GILBERT, O. L. & ANDERSON, P. (1998) *Habitat creation and repair,* Oxford University Press, Oxford.

GOODMAN, G. T. (1974) Ecology and the problems of rehabilitating wastes from mineral extraction. *Proceedings of the Royal Society of London.* , **339,** 373-387.

GOTELLI, N. J. & GRAVES, G. R. (1996) *Null models in ecology,* Smithsonian Institution Press, Washington.

GOULSON, D. (2003a) *Bumblebees: their behaviour and ecology,* Oxford Univiversity Press, Oxford.

GOULSON, D. (2003b) Conserving wild bees for crop pollination. *Food, Agriculture & Environment,* **1** 142 - 144.

GOULSON, D. (2003c) Effects of introduced bees on native ecosystems. *Annual Review of Ecology, Evolution, and Systematics,* **34,** 1-26.

GOULSON, D., HANLEY, M. E., DARVILL, B., ELLIS, J. S. & KNIGHT, M. E. (2005) Causes of rarity in bumblebees. *Biological Conservation,* **122,** 1-8.

GOULSON, D., HUGHES, W., DERWENT, L. & STOUT, J. (2002) Colony growth of the bumblebee, *Bombus terrestris*, in improved and conventional agricultural and suburban habitats. *Oecologia,* **130,** 267-273.

GOULSON, D. & STOUT, J. C. (2001) Homing ability of the bumblebee Bombus terrestris (Hymenoptera : Apidae). *Apidologie,* **32,** 105-111.

GOVERDE, M., SCHWEIZER, K., BAUR, B. & ERHARDT, A. (2002) Small-scale habitat fragmentation effects on pollinator behaviour: experimental evidence from the bumblebee *Bombus veteranus* on calcareous grasslands. *Biological Conservation,* **104,** 293-299.

GREEN, B. H. (1989) Agricultural impacts on the rural environment. *The Journal of Applied Ecology,* **26,** 793-802.

GREENLEAF, S., WILLIAMS, N., WINFREE, R. & KREMEN, C. (2007) Bee foraging ranges and their relationship to body size. *Oecologia,* **153,** 589-596.

GREENLEAF, S. S. & KREMEN, C. (2006) Wild bee species increase tomato production and respond differently to surrounding land use in Northern California. *Biological Conservation,* **133,** 81-87.

GREGORY, A. S. & VICKERS, A. W. (2003) Effects of amendments on soil structural development in a clay soil-forming material used as a cap for landfill restoration. *Soil Use and Management,* **19,** 273-276.

GRIMM, M. & BACKX, J. (1990) The restoration of shallow eutrophic lakes, and the role of northern pike, aquatic vegetation and nutrient concentration. *Hydrobiologia,* **200/201,** 557-566.

GRIXTI, J. C., WONG, L. T., CAMERON, S. A. & FAVRET, C. (2009) Decline of bumble bees (Bombus) in the North American Midwest. *Biological Conservation,* **142,** 75-84.

GUIMARÃES, P. R., DE AGUIAR, M. A. M., BASCOMPTE, J., JORDANO, P. & DOS REIS, S. F. (2005) Random initial condition in small Barabasi-Albert networks and deviations from the scale-free behavior. *Physical Review Letters,* **71,** 037101.

GUIMARÃES, P. R. & GUIMARÃES, P. (2006) Improving the analyses of nestedness for large sets of matrices. *Environmental Modelling & Software,* **21,** 1512-1513.

GUIMARAES, P. R., RICO-GRAY, V., DOS REIS, S. F. & THOMPSON, J. N. (2006) Asymmetries in specialization in ant–plant mutualistic networks. *Proceedings of the Royal Society: Biological Sciences,* **273,** 2041.

HANDEL, S. N., ROBINSON, G. R. & BEATTIE, A. J. (1994) Biodiversity resources for restoration ecology. *Restoration Ecology,* **2,** 230-241.

HANNON, L. E. & SISK, T. D. (2009) Hedgerows in an agri-natural landscape: Potential habitat value for native bees. *Biological Conservation,* **142,** 2140-2154

HARDER, L. D. (1985) Morphology as a predictor of flower choice by bumble bees. *Ecology,* **66,** 198-210.

HARDER, L. D. & CRUZAN, M. B. (1990) An evaluation of the physiological and evolutionary influences of inflorescence size and flower depth on nectar production. *Functional Ecology,* **4,** 559-572.

HARTUNG, S. C. & BRAWN, J. D. (2005) Effects of savanna restoration on the foraging ecology of insectivorous songbirds *The Condor,* **107,** 879-888.

HATFIELD, R. G. & LEBUHN, G. (2007) Patch and landscape factors shape community assemblage of bumble bees, Bombus spp. (Hymenoptera: Apidae), in montane meadows. *Biological Conservation,* **139,** 150-158.

HEGLAND, S. J. & BOEKE, L. (2006) Relationships between the density and diversity of floral resources and flower visitor activity in a temperate grassland community. *Ecological Entomology,* **31,** 532–538.

HEGLAND, S. J. & TOTLAND, Ø. (2005) Relationships between species' floral traits and pollinator visitation in a temperate grassland. *Oecologia,* **145,** 586-594.

HENLE, K., ALARD, D., CLITHEROW, J., COBB, P., FIRBANK, L., KULL, T., MCCRACKEN, D., MORITZ, R. F. A., NIEMELA, J., REBANE, M., WASCHER, D., WATT, A. &

YOUNG, J. (2008) Identifying and managing the conflicts between agriculture and biodiversity conservation in Europe-A review. *Agriculture, Ecosystems & Environment,* **124,** 60-71

HERRERA, J. (1985) Biologia reproductiva del matorral de Donana. *Ph.D diss.*, University of Seville, Seville.

HMSO (2002) *The landfill (England and Wales) regulations 2002,* Her Majesty's Stationery Office, London.

HOBBS, R. J. & HARRIS, J. A. (2001) Restoration ecology: Repairing the earth's ecosystems in the new millennium. *Restoration Ecology,* **9,** 239-246.

HOLL, K. D. (1995) Nectar resources and their influence on butterfly communities on reclaimed coal surface mines. *Restoration Ecology,* **3,** 76-85.

HOLLEY, K. & PHILLIPS, P. (1996) Planting trees on landfill. *IWM Proceedings.*

HOLT, R. D., KNIGHT, T. M. & BARFIELD, M. (2004) Allee effects, immigration, and the evolution of species' niches. *American Naturalist,* **163,** 253-262.

HOLZSCHUH, A., STEFFAN-DEWENTER, I., KLEIJN, D. & TSCHARNTKE, T. (2007) Diversity of flower-visiting bees in cereal fields: effects of farming system, landscape composition and regional context. *Journal of Applied Ecology,* **44,** 41-49.

HOOK, T. V. (1997) Insect coloration and implications for conservation. *The Florida Entomologist,* **80,** 193-210.

HOPWOOD, J. L. (2008) The contribution of roadside grassland restorations to native bee conservation. *Biological Conservation,* **114,** 2632-2640

HUTCHINGS, M. J. & STEWART, A. J. A. (2002) Calcareous grasslands. In: *Handbook of restoration ecology. Volume 2.* (Eds.) PERROW, M. R. & DAVY, A. J. Cambridge university press, Cambridge.

HUTCHINGS, T. R., SINNETT, D., PEACE, A. J. & MOFFAT, A. J. (2006) The effect of woodland growth on a containment landfill site in Hertfordshire, UK. *Urban Forestry & Urban Greening,* **5,** 169-176.

INOUYE, D. W. (2007) The value of bees. *Biological Conservation,* **140,** 198-199

IRELAND, E. M. (1991) Factors influencing the establishment of floristically rich grasslands on a restored landfill site. *PhD Thesis*, Essex University Colchester

JONES, G. H., TRUEMAN, I. C. & MILLETT, P. (1995) The use of hay strewing to create species-rich grasslands (i) general principles and hay strewing versus seed mixes. *Land Contamination and Reclamation,* **3,** 104–107.

JORDANO, P. (1987) Patterns of mutualistic interactions in pollination and seed dispersal: Connectance, dependence asymmetries, and coevolution. *The American Naturalist,* **129,** 657-677.

JORDANO, P., BASCOMPTE, J. & OLESEN, J. M. (2003) Invariant properties in coevolutionary networks of plant-animal interactions. *Ecology Letters,* **6,** 69-81.

JORDANO, P., BASCOMPTE, J. & OLESEN, J. M. (2006) The ecological consequences of complex topology and nested structure in pollination webs. In: *Plant–pollinator interactions: from specialization to generalization.* (Eds.) WASER, N. M. & OLLERTON, J. University of Chicago Press, Chicago.

KAISER-BUNBURY, C. N., MEMMOTT, J. & MÜLLER, C. B. (2009) Community structure of pollination webs of Mauritian heathland habitats. *Perspectives in Plant Ecology, Evolution and Systematics,* **In Press, Corrected Proof**.

KALIKHMAN, I. (2007) Patchy distribution fields: A spiral survey design and reconstruction adequacy. *Environmental Monitoring and Assessment,* **124,** 243-252.

KASINA, J. M., MBURU, J., KRAEMER, M. & HOLM-MUELLER, K. (2009) Economic benefit of crop pollination by bees: A case of Kakamega small-holder farming in Western Kenya. *Journal of Economic Entomology,* **102,** 467-473.

KEARNS, C. A. & INOUYE, D. W. (1993) *Techniques for pollination biologists,* University press of Colorado, Colorado.

KEARNS, C. A. & INOUYE, D. W. (1997) Pollinators, flowering plants, and conservation biology. *BioScience,* **47,** 297-307.

KEARNS, C. A., INOUYE, D. W. & WASER, N. M. (1998) Endangered mutualisms: The conservation of plant-pollinator interactions. *Annual Review of Ecology and Systematics,* **29,** 83-112.

KEITH, S. A., NEWTON, A. C., MORECROFT, M. D., BEALEY, C. E. & BULLOCK, J. M. (2009) Taxonomic homogenization of woodland plant communities over 70 years. *Proceedings of the Royal Society B: Biological Sciences,* **276,** 3539-3544.

KELLS, A. R. & GOULSON, D. (2003) Preferred nesting sites of bumblebee queens (*Hymenoptera: Apidae*) in agroecosystems in the UK. *Biological Conservation,* **109,** 165-174.

KENT, D. H. (1992) *List of vascular plants of the British Isles,* BSBI Publications, London.

KESSELL, S. R. & WHITTAKER, R. H. (1976) Comparisons of three ordination techniques. *Plant Ecology,* **32,** 21-29.

KEVAN, P. G. & BAKER, H. G. (1983) Insects as flower visitors and pollinators. *Annual Review of Entomology,* **28,** 407-453.

KEVAN, P. G., IMPERATRIZ-FONSECA, V., FRANKIE, G. W., O'TOOLE, C., JONES, R. & VERGARA, C. H. (2002) The conservation link between agriculture and nature. *Ministry of Environment, Government of Brazil, Brasilia.* , 313.

KIEHL, K. & PFADENHAUER, J. (2007) Establishment and persistence of target species in newly created calcareous grasslands on former arable fields. *Plant Ecology,* **189,** 31-48.

KIM, K., LEE, E. & CHO, K.-H. (2004) The plant community of Nanjido, a representative nonsanitary landfill in South Korea: Implications for restoration alternatives. *Water, Air, & Soil Pollution,* **154,** 167-185.

KLEEM, M. (1996) Man-made bee habitats in the anthropogenous landscape of central Europe - substitutes for threatened or destroyed riverine habitats. In: *The conservation of bees.* (Eds.) MATHESON, A., BUCHMANN, S. L., O'TOOLE, C., WESTRICH, P. & WILLIAMS, I. H. Academic Press, London.

KLEIJN, D. & VAN LANGEVELDE, F. (2006) Interacting effects of landscape context and habitat quality on flower visiting insects in agricultural landscapes. *Basic and Applied Ecology,* **7,** 201-214.

KLEIN, A.-M., VAISSIÈRE, B. E., CANE, J. H., STEFFAN-DEWENTER, I., CUNNINGHAM, S. A., KREMEN, C. & TSCHARNTKE, T. (2007) Importance of pollinators in changing landscapes for world crops. *Proceedings of the Royal Society B: Biological Sciences,* **274,** 303-313.

KLEIN, A. M., STEFFAN-DEWENTER, I. & TSCHARNTKE, T. (2003) Pollination of *Coffea canephora* in relation to local and regional agroforestry management. *The Journal of Applied Ecology,* **40,** 837-845.

KNIGHT, M. E., MARTIN, A. P., BISHOP, S., OSBORNE, J. L., HALE, R. J., SANDERSON, R. A. & GOULSON, D. (2005) An interspecific comparison of foraging range and nest density of four bumblebee (Bombus) species. *Molecular Ecology,* **14,** 1811-1820.

KNUTH, P. (1906-1909) *Handbook of flower pollination Vols. I - III* Clarendon press, Oxford.

KOSIOR, A., CELARY, W., OLEJNICZAK, P., FIJAL, J., KRÓL, W., SOLARZ, W. & PLONKA, P. (2007) The decline of the bumble bees and cuckoo bees (Hymenoptera: Apidae: Bombini) of Western and Central Europe. *Oryx,* **41,** 79-88.

KRAUSS, J., STEFFAN-DEWENTER, I. & TSCHARNTKE, T. (2003) How does landscape context contribute to effects of habitat fragmentation on diversity and population density of butterflies? *Journal of Biogeography,* **30,** 889-900.

KREMEN, C. (1992) Assessing the indicator properties of species assemblages for natural areas monitoring. *Ecological Applications,* **2,** 203-217.

KREMEN, C. & RICKETTS, T. (2000) Global perspectives on pollination disruptions. *Conservation Biology,* **14,** 1226-1228.

KREMEN, C., WILLIAMS, N. M., BUGG, R. L., FAY, J. P. & THORP, R. W. (2004) The area requirements of an ecosystem service: crop pollination by native bee communities in California. *Ecology Letters,* **7,** 1109-1119.

KREYER, D., OED, A., WALTHER-HELLWIG, K. & FRANKL, R. (2004) Are forests potential landscape barriers for foraging bumblebees? Landscape scale experiments with *Bombus terrestris* agg. and *Bombus pascuorum* (Hymenoptera, Apidae). *Biological Conservation,* **116,** 111-118.

KWAK, M. M. (1994) Planten en bestuivers: Achteruitgang leidt tot verschuivende relaties. *Landschap,* **11,** 29-39.

KWAK, M. M., VELTEROP, O. & VAN ANDEL, J. (1998) Pollen and gene flow in fragmented habitats. *Applied Vegetation Science,* **1,** 37-54.

LAGERLOF, J., STARK, J. & SVENSSON, B. (1992) Margins of agricultural fields as habitats for pollinating insects. *Agriculture, Ecosystems & Environment,* **40,** 117-124.

LANZA, J., SMITH, G. C., SACK, S. & CASH, A. (1995) Variation in nectar volume and composition of *Impatiens capensis* at the individual, plant, and population levels. *Oecologia,* **102,** 113-119.

LEGENDRE, P. & LEGENDRE, L. (1998) *Numerical ecology,* Elsevier Scientific, Amsterdam.

LEHMANN, C. & REBELE, F. (2002) Successful Management of *Calcalmagrostis epigejos* (l.) Roth on a Sabdy Landfill Site. *Jounal of Applied Botany,* **76,** 77 - 81.

LEWINSOHN, T. M., INACIO PRADO, P., JORDANO, P., BASCOMPTE, J. & M. OLESEN, J. (2006) Structure in plant-animal interaction assemblages. *Oikos,* **113,** 174-184.

LINSLEY, E. G. (1958) The ecology of solitary bees. *Hilgardia,* **27,** 543-599.

LUNDHOLM, J. T. & LARSON, D. W. (2003) Relationships between spatial environmental heterogeneity and plant species diversity on a limestone pavement. *Ecography,* **26,** 715-722.

LYE, G., PARK, K., OSBORNE, J., HOLLAND, J. & GOULSON, D. (2009) Assessing the value of rural stewardship schemes for providing foraging resources and nesting habitat for bumblebee queens (Hymenoptera: Apidae). *Biological Conservation,* **142,** 2023-2032

MAAREL, E. V. D. (Ed.) (2005) *Vegetation Ecology,* Blackwell Publishing, London.

MACCHERINI, S., BACARO, G., FAVILLI, L., PIAZZINI, S., SANTI, E. & MARIGNANI, M. (2009) Congruence among vascular plants and butterflies in the evaluation of grassland restoration success. *Acta Oecologica,* **35,** 311-317.

MAKEL ENGINEERING (2006) Landfill Diagram Accessed; October 2006. From: http://www.makelengineering.com/dir/Technologies/Power/Images/Landfill.jpg

MARSHALL, E. J. P. & MOONEN, A. C. (1998) *A review of field margin conservation strips in Europe,* Long Ashton Research Station.

MARTINEZ, N. D., HAWKINS, B. A., DAWAH, H. A. & FEIFAREK, B. P. (1999) Effects of sampling effort on characterization of food-web structure. . *Ecology,* **80,** 1044-1055.

MAS (2008a) Agricultural mix: Long term 5 years plus. Accessed; November 2008. From: http://www.meadowmania.co.uk/default.cfm/loaddoc.97

MAS (2008b) Wild Flower Seed mixture for calcareous soils. Accessed; November 2008. From: http://www.meadowmania.co.uk/default.cfm/loaddoc.51

McCune, B. & Grace, J. B. (2002) *Analysis of ecological communities,* MjM Software Gleneden Beach, Oregon.

McFrederick, Q. S. & LeBuhn, G. (2006) Are urban parks refuges for bumble bees Bombus spp. (Hymenoptera: Apidae)? *Biological Conservation,* **129,** 372-382.

McMaster, R. T. (2005) Factors influencing vascular plant diversity on 22 islands off the coast of eastern North America. *Journal of Biogeography,* **32,** 475-492.

Memmott, J. (1999) The structure of a plant-pollinator food web. *Ecology Letters,* **2,** 276-280.

Memmott, J., Waser, N. M. & Price, M. V. (2004) Tolerance of pollination networks to species extinctions. *Proceedings of the Royal Society B: Biological Sciences,* **271,** 2605–2611.

Met. Office (2007) Summer 2007 - a wet season Accessed; July 2009. From: http://www.metoffice.gov.uk/corporate/pressoffice/2007/pr20070831.html

Met. Office (2008) Summer 2008 Accessed; July 2009. From: http://www.metoffice.gov.uk/climate/uk/2008/summer.html

Met. Office (2009) Statement on summer forecast 2009 Accessed; July 2009. From: http://www.metoffice.gov.uk/corporate/pressoffice/2009/pr20090729.html

Meyer, B., Jauker, F. & Steffan-Dewenter, I. (2009) Contrasting resource-dependent responses of hoverfly richness and density to landscape structure. *Basic and Applied Ecology,* **10,** 178-186.

Michener, C. D. (2007) *The bees of the world* Johns Hopkins University Press, Baltimore.

Mitchley, J., Buckley, G. P. & Helliwell, D. R. (1996) Vegetation establishment on chalk marl spoil: The role of nurse grass species and fertiliser application. *Journal of Vegetation Science,* **7,** 543-548.

Moffat, A. J. & Houston, T. J. (1991) Tree establishment and growth at Pitsea landfill site, Essex, U. K. *Waste Management & Research,* **9,** 35-46.

Montalvo, A. M., Rice, S. L. W., Buchmann, S. L., Cory, C., Handel, S. N., Nabhan, G. P., Primack, R. & Robichaux, R. H. (1997) Restoration biology: A population biology perspective. *Restoration Ecology,* **5,** 277-290.

MORANDIN, L. A. & WINSTON, M. L. (2006) Pollinators provide economic incentive to preserve natural land in agro-ecosystems. *Agriculture, Ecosystems & Environment,* **116,** 289-292.

MORANDIN, L. A., WINSTON, M. L., ABBOTT, V. A. & FRANKLIN, M. T. (2007) Can pastureland increase wild bee abundance in agriculturally intense areas? *Basic and Applied Ecology,* **8,** 117-124

MORRIS, M. G. (1969) Populations of invertebrate animals and the management of chalk grassland in Britain. *Biological Conservation,* **1,** 225-231.

MORTIMER, S. R., HOLLIER, J. A. & BROWN, V. K. (1998) Interactions between plant and insect diversity in the restoration of lowland calcareous grasslands in southern Britain. *Applied Vegetation Science,* **1,** 101-114.

MUNGUIRA, M. L. & THOMAS, J. A. (1992) Use of road verges by butterfly and burnet populations, and the effect of roads on adult dispersal and mortality. *Journal of applied ecology,* **29,** 316-329.

MURDOCH, W. W., EVANS, F. C. & PETERSON, C. H. (1972) Diversity and pattern in plants and insects. *Ecology,* **53,** 819-829.

NATURAL ENGLAND (2006) Nature on the map Accessed; October 2006. From: http://www.natureonthemap.org.uk/map.aspx

NATURAL ENGLAND (2009) What are Local Nature Reserves? Accessed; June 2009. From: http://www.lnr.naturalengland.org.uk/special/lnr/office.htm

NEAL, P. R. (1998) Pollinator restoration. *Trends in Ecology & Evolution,* **13,** 132-133.

NICHOLS, O. G. & NICHOLS, F. M. (2003) Long-term trends in faunal recolonization after bauxite mining in the Jarrah forest of Southwestern Australia. *Restoration Ecology,* **11,** 261-272.

NICKY'S NURSERY (2008) D.O.T. Verge Mix WS-GPS07 Accessed; November 2008. From: http://www.nickys-nursery.co.uk/seeds/wholesale/grassseed3.htm

NIELSEN, A. & BASCOMPTE, J. (2007) Ecological networks, nestedness and sampling effort. *Journal of Ecology,* **95,** 1134-1141.

NOORDIJK, J., DELILLE, K., SCHAFFERS, A. P. & SÝKORA, K. V. (2009) Optimizing grassland management for flower-visiting insects in roadside verges. *Biological Conservation,* **142,** 2097-2103

ÖCKINGER, E. & SMITH, H. G. (2007) Semi-natural grasslands as population sources for pollinating insects in agricultural landscapes. *Journal of Applied Ecology,* **44,** 50-59.

OLESEN, J. M., BASCOMPTE, J., DUPONT, Y. L. & JORDANO, P. (2007) The modularity of pollination networks. *Proceedings of the National Academy of Sciences,* **104,** 19891-19896.

OLESEN, J. M. & JORDANO, P. (2002) Geographic patterns in plant-pollinator mutualistic networks. *Ecology,* **83,** 2416-2424.

OLLERTON, J. (1999) The evolution of pollinator-plant relationships within the arthropods. In: *Evolution and Phylogeny of the Arthropoda.* (Eds.) MELIC, A., DEHARO, J. J., MENDEZ, M. & RIBERA, I. Zaragoza, Aragon.

OLLERTON, J., ALARCÓN, R., WASER, N. M., PRICE, M. V., WATTS, S., CRANMER, L., HINGSTON, A., PETER, C. I. & ROTENBERRY, J. (2009) A global test of the pollination syndrome hypothesis. *Annals of Botany,* **103,** 1471-1480.

OLLERTON, J., JOHNSON, S. D., CRANMER, L. & KELLIE, S. (2003) The pollination ecology of an assemblage of grassland asclepiads in South Africa. *Ann Bot,* **92,** 807-834.

OLLERTON, J., KILLICK, A., LAMBORN, E., WATTS, S. & WHISTON, M. (2007) Multiple meanings and modes: on the many ways to be a generalist flower. *Taxon,* **56,** 717-728.

OLSSON, O., BROWN, J. S. & SMITH, H. G. (2002) Long- and short-term state-dependent foraging under predation risk: an indication of habitat quality. *Animal Behaviour,* **63,** 981-989.

OPPERMAN, J. J. & MERENLENDER, A. M. (2000) Deer herbivory as an ecological constraint to restoration of degraded riparian corridors. *Restoration Ecology,* **8,** 41-47.

ORTH, R. J., LUCKENBACH, M. & MOORE, K. A. (1994) Seed dispersal in a marine macrophyte: Implications for colonization and restoration. *Ecology,* **75,** 1927-1939.

OSBORNE, J., L., MARTIN, A., P., SHORTALL, C., R. , TODD, A., D. , GOULSON, D., KNIGHT, M., E. , HALE, R., J. & SANDERSON, R., A. (2008a) Quantifying and comparing bumblebee nest densities in gardens and countryside habitats. *Journal of Applied Ecology,* **45,** 784-792.

OSBORNE, J., MARTIN, A., CARRECK, N., SWAIN, J., KNIGHT, M., GOULSON, D., HALE, R. & SANDERSON, R. (2008b) Bumblebee flight distances in relation to the forage landscape. *Journal of Animal Ecology,* **77,** 406-415.

OSBORNE, J. L., CLARK, S. J., MORRIS, R. J., WILLIAMS, I. H., RILEY, J. R., SMITH, A. D., REYNOLDS, D. R. & EDWARDS, A. S. (1999) A landscape-scale study of bumble bee foraging range and constancy, using harmonic radar. *Journal of Applied Ecology,* **36,** 519-533.

OSBORNE, J. L. & CORBET, S. A. (1994) Managing habitats for pollinators in farmland. *Aspects of Applied Biology (United Kingdom); Arable farming under CAP reform*, 207-215.

OSBORNE, J. L. & WILLIAMS, I. H. (1996) Bumble bees as pollinators of crops and wild flowers. *Bumble bees for pleasure and profit. IBRA, Cardiff*, 24–32.

PACINI, E., NEPI, M. & VESPRINI, J. L. (2003) Nectar biodiversity: a short review. *Plant Systematics and Evolution,* **238,** 7-21.

PALMER, M. A., AMBROSE, R. F. & POFF, N. L. (1997) Ecological theory and community restoration ecology. *Restoration Ecology,* **5,** 291-300.

PAUSAS, J. G. & AUSTIN, M. P. (2001) Patterns of plant species richness in relation to different environments: An appraisal. *Journal of Vegetation Science,* **12,** 153-166.

PAUW, A. (2007) Collapse of a pollination web in small conservation areas. *Ecology,* **88,** 1759–1769.

PERCIVAL, M. S. (1961) Types of nectar in angiosperms. *New Phytologist,* **60,** 235-281.

PETANIDOU, T., KALLIMANIS, A. S., TZANOPOULOS, J., SGARDELIS, S. P. & PANTIS, J. D. (2008) Long-term observation of a pollination network: fluctuation in species and interactions, relative invariance of network structure and implications for estimates of specialization. *Ecology Letters,* **11,** 564-575.

PLEASANTS, J. M. & CHAPLIN, S. J. (1983) Nectar production rates of *Asclepias quadrifolia*: causes and consequences of individual variation. *Oecologia,* **59,** 232-238.

POLLARD, E. & YATES, T. J. (1993) *Monitoring butterflies for ecology and conservation,* Chapman & Hall, London, UK.

POLLINATION GUELPH (2009) World's first large-scale pollinator park. Accessed; June 2009. From: http://www.pollinator.ca/guelph/

POTTS, S., G. , VULLIAMY, B., DAFNI, A., NE'EMAN, G. & WILLMER, P. (2003a) Linking bees and flowers: How do floral communities structure pollinator communities? *Ecology,* **84,** 2628-2642.

POTTS, S. G., PETANIDOU, T., ROBERTS, S., O'TOOLE, C., HULBERT, A. & WILLMER, P. (2006) Plant-pollinator biodiversity and pollination services in a complex Mediterranean landscape. *Biological Conservation,* **129,** 519-529.

POTTS, S. G., VULLIAMY, B., DAFNI, A., NE'EMAN, G., O'TOOLE, C., ROBERTS, S. & WILLMER, P. (2003b) Response of plant-pollinator communities to fire: changes in diversity, abundance and floral reward structure. *Oikos,* **101,** 103-112.

POTTS, S. G., VULLIAMY, B., ROBERTS, S., O'TOOLE, C., DAFNI, A., NE'EMAN, G. & WILLMER, P. G. (2004) Nectar resource diversity organises flower-visitor community structure. *Entomologia Experimentalis et Applicata,* **113,** 103-107.

POTTS, S. G., WOODCOCK, B. A., ROBERTS, S. P. M., TSCHEULIN, T., PILGRIM, E. S., BROWN, V. K. & TALLOWIN, J. R. (2009) Enhancing pollinator biodiversity in intensive grasslands. *Journal of Applied Ecology,* **46,** 369-379.

POWELL, A. H. & POWELL, G. V. N. (1987) Population dynamics of male Euglossine bees in Amazonian forest fragments. *Biotropica,* **19,** 176-179.

PRIMACK, R. B. (1983) Insect pollination in the New Zealand mountain flora. *New Zealand Journal of Botany,* **21,** 317-333.

PRYS-JONES, O. E. & CORBET, S. A. (1991) *Bumblebees,* Richmond Publishing Co. Ltd. , Slough.

PYWELL, R. F., WARMAN, E. A., CARVELL, C., SPARKS, T. H., DICKS, L. V., BENNETT, D., WRIGHT, A., CRITCHLEY, C. N. R. & SHERWOOD, A. (2005) Providing foraging resources for bumblebees in intensively farmed landscapes. *Biological Conservation,* **121,** 479-494.

RAHMAN, L. (2009) The potential of restored landfill sites for biodiversity conservation *PhD Applied science.* University of Northampton, Northampton UK.

RAWLINSON, H., DICKINSON, N., NOLAN, P. & PUTWAIN, P. (2004) Woodland establishment on closed old-style landfill sites in North West England. *Forest Ecology and Management,* **202,** 265-280.

REBELE, F. & LEHMANN, C. (2002) Restoration of a landfill site in Berlin, Germany by spontaneous and direct succession. *Restoration Ecology,* **10,** 340-347.

REED, C. (1995) Insects surveyed on flowers in native and reconstructed prairies (Minnesota). *Restoration and Management Notes,* **13,** 210-213.

REMON, E., BOUCHARDON, J. L., CORNIER, B., GUY, B., LECLERC, J. C. & FAURE, O. (2005) Soil characteristics, heavy metal availability and vegetation recovery at a former metallurgical landfill: Implications in risk assessment and site restoration. *Environmental Pollution,* **137,** 316-323.

REYNOLDSON, T. B., NORRIS, R. H., RESH, V. H., DAY, K. E. & ROSENBERG, D. M. (1997) The reference condition: A comparison of multimetric and multivariate approaches to assess water-quality impairment using benthic macroinvertebrates. *Journal of the North American Benthological Society,* **16,** 833-852.

RHOADES, C. C., ECKERT, G. E. & COLEMAN, D. C. (1998) Effect of pasture trees on soil nitrogen and organic matter: Implications for tropical montane forest restoration. *Restoration Ecology,* **6,** 262-270.

RICKETTS, T. H. (2001) The matrix matters: effective isolation in fragmented landscapes. *The American Naturalist,* **158,** 87-99.

RICKETTS, T. H., REGETZ, J., STEFFAN-DEWENTER, I., CUNNINGHAM, S. A., KREMEN, C., BOGDANSKI, A., GEMMILL-HERREN, B., GREENLEAF, S. S., KLEIN, A. M., MAYFIELD, M. M., MORANDIN, L. A., OCHIENG, A. & VIANA, B. F. (2008) Landscape effects on crop pollination services: are there general patterns? *Ecology Letters,* **11,** 499-515.

ROBINSON, G. R. & HANDEL, S. N. (1993) Forest restoration on a closed landfill: rapid addition of new species by bird dispersal. *Conservation Biology,* **7,** 271-278.

ROSE, F. & O'REILLY, C. (2006) *The Wild Flower Key - How to identify wild flowers, trees and shrubs in Britain and Ireland,* Penguin Group, London.

ROTHROCK, J. T. (1867) The fertilization of flowering plants. *The American Naturalist,* **1,** 64-72.

ROWELL, D. L. (1994) *Soil science: Methods & Applications,* Longman Scientific and Technical, Harlow - UK.

RUHREN, S. & HANDEL, S. N. (2003) Herbivory constrains survival, reproduction and mutualisms when restoring nine temperate forest herbs. *Journal of the Torrey Botanical Society,* **130,** 34-42.

RUIZ-JAEN, M. C. & MITCHELL AIDE, T. (2005) Restoration success: How is it being measured? *Restoration Ecology,* **13,** 569-577.

SAMWAYS, M. J. (1992) Some comparative insect conservation issues of north temperate, tropical, and south temperate landscapes. *Agriculture, Ecosystems & Environment,* **40,** 137-154.

SANDERSON, M. A., ROTZ, C. A., FULTZ, S. W. & RAYBURN, E. B. (2001) Estimating forage mass with a commercial capacitance meter, rising plate meter, and pasture ruler. *Agron J,* **93,** 1281-1286.

SANTAMARIA, L. & RODRIGUEZ-GIRONES, M. A. (2007) Linkage rules for plant–pollinator networks: Trait complementarity or exploitation barriers? *PLoS Biol,* **5,** 354 - 362.

SCHILTHUIZEN, M. (2008) *The loom of life: unravelling ecosystems,* Springer, Leiden, The Netherlands.

SERI (2004) Society for Ecological Restoration International - Science & Policy Working Group. *SERI Primer on Ecological Restoration,* Accessed: May 2008. From: http://www.ser.org/content/ecological_restoration_primer.asp

SHARP, M. A., PARKS, D. R. & EHRLICH, P. R. (1974) Plant resources and butterfly habitat selection. *Ecology,* **55,** 870-875.

SHARROW, S. H. (1984) A simple disc meter for measurement of pasture height and forage bulk. *Journal of Range Management,* **37,** 94-95.

SHEPERD, M., BUCHMANN, S. L., VAUGHAN, M. & BLACK, S. H. (2003) *Pollinator conservation handbook : A guide to understanding, protecting and providing habitat for native pollinator insects,* Xerces Society, Portland, USA.

SIMBERLOFF, D. S. & WILSON, E. O. (1969) Experimental zoogeography of Islands: The colonization of empty Islands. *Ecology,* **50,** 278-296.

SIMMONS, E. (1999) Restoration of landfill sites for ecological diversity. *Waste Management and Research,* **17,** 511-519.

SJÖDIN, N. (2007) Pollinator behavioural responses to grazing intensity. *Biodiversity and Conservation,* **16,** 2103-2121.

SJÖDIN, N., BENGTSSON, J. & EKBOM, B. (2008) The influence of grazing intensity and landscape composition on the diversity and abundance of flower-visiting insects. *Journal of Applied Ecology,* **45,** 763-772.

SMART, S. M., FIRBANK, L. G., BUNCE, R. G. H. & WATKINS, J. W. (2000) Quantifying changes in abundance of food plants for butterfly larvae and farmland birds. *Journal of Applied Ecology,* **37,** 398-414.

SPSS (2003) SPSS version 11.5 for Windows. Chicago, SPSS Inc.

STACE, C. (1991) *New flora of the British Isles,* Cambridge University Press, Cambridge.

STANG, M., KLINKHAMER, P. & VAN DER MEIJDEN, E. (2007) Asymmetric specialization and extinction risk in plant–flower visitor webs: a matter of morphology or abundance? *Oecologia,* **151,** 442-453.

STANG, M., KLINKHAMER, P. G. L. & VAN DER MEIJDEN, E. (2006) Size constraints and flower abundance determine the number of interactions in a plant-flower visitor web. *OIKOS* **112,** 111-121.

STEFFAN-DEWENTER, I. (2003) Importance of habitat area and landscape context for species richness of bees and wasps in fragmented orchard meadows. *Conservation Biology,* **17,** 1036-1044.

STEFFAN-DEWENTER, I., MUNZENBERG, U., BURGER, C., THIES, C. & TSCHARNTKE, T. (2002) Scale-dependent effects of landscape context on three pollinator guilds. *Ecology,* **83,** 1421-1432.

STEFFAN-DEWENTER, I., POTTS, S. G. & PACKER, L. (2005) Pollinator diversity and crop pollination services are at risk. *Trends in Ecology & Evolution,* **20,** 651-652.

STEFFAN-DEWENTER, I. & TSCHARNTKE, T. (1999) Effects of habitat isolation on pollinator communities and seed set. *Oecologia,* **121,** 432 - 440.

STEFFAN-DEWENTER, I. & TSCHARNTKE, T. (2000) Butterfly community structure in fragmented habitats. *Ecology Letters,* **3,** 449-456.

STEFFAN-DEWENTER, I. & TSCHARNTKE, T. (2001) Succession of bee communities on fallows. *Ecography,* **24,** 83-93.

STEFFAN-DEWENTER, I. & TSCHARNTKE, T. (2002) Insect communities and biotic interactions on fragmented calcareous grasslands: a mini review. *Biological Conservation,* **104,** 275-284.

STILES, D. (2008) War on waste. Accessed; June 2009. From: http://kn.theiet.org/magazine/issues/0820/war-waste-0820.cfm

STOUT, J. C. (2007) Pollination of invasive *Rhododendron ponticum* (Ericaceae) in Ireland. *Apidologie,* **38,** 198-206.

STREEVER, B., ZEDLER, J., MITSCH, W. J., WU, X. B., NAIRN, R. W. & WANG, N. (2000) To plant or not to plant. *BioScience,* **50,** 188-190.

STRONG, D. R., LAWTON, J. H. & SOUTHWOOD, R. (1984) *Insects on plants,* Blackwell Scientific Publications, Oxford.

STUBBS, A. E. & FALK, S. J. (2000) *British hoverflies: An illustrated identification guide.,* British Entomological & Natural History Society, Reading.

SWEENEY, B. W., CZAPKA, S. J. & YERKES, T. (2002) Riparian forest restoration: Increasing success by reducing plant competition and herbivory. *Restoration Ecology,* **10,** 392-400.

TARRANT, S. & OLLERTON, J. (In prep.) Probing interaction network structure : Experimental core floral resource removal and its effects upon pollinating insects

TEPEDINO, V. J. & STANTON, N. L. (1982) Estimating floral resources and flower visitors in studies of pollinator-plant communities. *Oikos,* **38,** 384-386.

THOMAS, J. A., BOURN, N. A. D., CLARKE, R. T., STEWART, K. E., SIMCOX, D. J., PEARMAN, G. S., CURTIS, R. & GOODGER, B. (2001) The quality and isolation of habitat patches both determine where butterflies persist in fragmented landscapes. *Proceedings of the Royal Society B: Biological Sciences,* **268,** 1791-1796.

THOMAS, J. A., TELFER, M. G., ROY, D. B., PRESTON, C. D., GREENWOOD, J. J. D., ASHER, J., FOX, R., CLARKE, R. T. & LAWTON, J. H. (2004) Comparative losses of British butterflies, birds, and plants and the global extinction crisis. *Science,* **303,** 1879-1881.

THOMAS, S. (2000) Progress on beetle banks in UK arable farming *Pesticide Outlook.*

TODD, J. E. (1879) On certain contrivances for cross-fertilization in flowers. *The American Naturalist,* **13,** 1-6.

TOFTS, R. (1999) *Cirsium eriophorum* (L.) Scop. (*Carduus eriophorus* L.; *Cnicus eriophorus* (L.) Roth). *Journal of Ecology,* **87,** 529-542.

TOMLINSON, D. & STILL, R. (2002) *Britain's butterflies,* Wild Guides, Old Basing, Hampshire.

TOSH, J. E., SENIOR, E., SMITH, J. E. & WATSON-CRAIK, I. A. (1994) Landfill site restoration: The inimical challenges of ethylene and methane. *Environmental Pollution,* **83,** 335-340.

TRELEASE, W. (1881) The fertilization of Scrophularia. *Bulletin of the Torrey Botanical Club,* **8,** 133-140.

TSCHARNTKE, T., GATHMANN, A. & STEFFAN-DEWENTER, I. (1998) Bioindication using trap-nesting bees and wasps and their natural enemies: community structure and interactions. *Journal of Applied Ecology,* **35,** 708-719.

TSCHARNTKE, T., KLEIN, A. M., KRUESS, A., STEFFAN-DEWENTER, I. & THIES, C. (2005) Landscape perspectives on agricultural intensification and biodiversity - ecosystem service management. *Ecology Letters,* **8,** 857-874.

TSCHARNTKE, T., STEFFAN-DEWENTER, I., KRUESS, A. & THIES, C. (2002) Characteristics of insect populations on habitat fragments: A mini Review. *Ecological Research,* **17,** 229 - 239.

TYLIANAKIS, J. M., KLEIN, A.-M. & TSCHARNTKE, T. (2005) Spatial variation in the diversity of Hymenoptera across a tropical habitat gradient. *Ecology,* **86,** 3296-3302.

U.N. (2006) World population prospects:The 2006 revision population database Accessed; September 2007. From: http://esa.un.org/unpp/

ULRICH, W., ALMEIDA-NETO, M. & GOTELLI, N. J. (2009) A consumer's guide to nestedness analysis. *Oikos,* **118,** 3-17.

UNEP (2005) *The millennium ecosystem assessement - Ecosystems and human well-being,* Island Press.

VÁZQUEZ, D. P. & AIZEN, M. A. (2006) Community-wide patterns of specialization in plant-pollinator interactions revealed by null models. In: *Plant–pollinator* *interactions: from specialization to generalization.* (Eds.) WASER, N. M. & OLLERTON, J. Chicago Press, Chicago.

VAZQUEZ, D. P., BLUTHGEN, N., CAGNOLO, L. & CHACOFF, N. P. (2009) Uniting pattern and process in plant-animal mutualistic networks: a review. *Annals of Botany,* **103,** 1445.

VULLIAMY, B., G. POTTS, S. & G. WILLMER, P. (2006) The effects of cattle grazing on plant-pollinator communities in a fragmented Mediterranean landscape. *Oikos,* **114,** 529-543.

WASER, N. M., CHITTKA, L., PRICE, M. V., WILLIAMS, N. M. & OLLERTON, J. (1996) Generalization in pollination systems, and why it matters. *Ecology,* **77,** 1043-1060.

WASER, N. M. & REAL, L. A. (1979) Effective mutualism between sequentially flowering plant species. *Nature,* **281,** 670-672.

WATSON, D. & HACK, V. (2000) *Wildlife management and habitat creation on landfill sites : a manual of best practice,* Ecoscope Applied Ecologists.

WENZEL, M., SCHMITT, T., WEITZEL, M. & SEITZ, A. (2006) The severe decline of butterflies on western German calcareous grasslands during the last 30 years: A conservation problem. *Biological Conservation,* **128,** 542-552.

WESTPHAL, C., STEFFAN-DEWENTER, I. & TSCHARNTKE, T. (2003) Mass flowering crops enhance pollinator densities at a landscape scale. *Ecology Letters,* **6,** 961-965.

WESTPHAL, C., STEFFAN-DEWENTER, I. & TSCHARNTKE, T. (2006) Bumblebees experience landscapes at different spatial scales: possible implications for coexistence. *Oecologia,* **149,** 289-300.

WESTRICH, P. (1996) Habitat requirements of central European bees and problems of partial habitats. In: *The conservation of bees.* (Eds.) MATHESON, A., BUCHMANN, S. L., O'TOOLE, C., WESTRICH, P. & WILLIAMS, I. H. Academic Press, London.

WILLIAMS, C. S. (1995) Conserving Europe's bees: why all the buzz? *Trends in Ecology & Evolution,* **10,** 309-310.

WILLIAMS, N. (2006) Gone, gone and going. *Current Biology,* **16,** R513-R515.

WILLIAMS, P. (1982) The distribution and decline of British bumble bees (Bombus Latr.). *Journal of Apicultural Research* **21,** 236-245.

WILLIAMS, P. (2005) Does specialization explain rarity and decline among British bumblebees? A response to Goulson et al. *Biological Conservation,* **122,** 33-43.

WILLIAMS, P. H., ARAUJO, M. B. & RASMONT, P. (2007) Can vulnerability among British bumblebee (Bombus) species be explained by niche position and breadth? *Biological Conservation,* **138,** 493-505.

WILLIAMS, P. H. & OSBORNE, J. L. (2009) Bumblebee vulnerability and conservation world-wide. *Apidologie,* **40,** 367-387.

WINFREE, R., AGUILAR, R., VÃ¡ZQUEZ, D. P., LEBUHN, G. & AIZEN, M. A. (2009) A meta-analysis of bees' responses to anthropogenic disturbance. *Ecology,* **90,** 2068-2076.

WINFREE, R., WILLIAMS, N. M., DUSHOFF, J. & KREMEN, C. (2007) Native bees provide insurance against ongoing honey bee losses. *Ecology Letters,* **10,** 1105-1113.

WITHGOTT, J. (1999) Pollination migrates to top of conservation agenda. *BioScience,* **49,** 857-862.

WONG, M. H. (1988) Soil and plant characteristics of landfill sites near Merseyside, England. *Environmental Management,* **12,** 491-499.

WUNDERLE, J. M. (1997) The role of animal seed dispersal in accelerating native forest regeneration on degraded tropical lands. *Forest Ecology and Management,* **99,** 223-235.

ZAHRADNIK, J. & SEVERA, F. (2000) *Bees and wasps,* Aventinum Publishing House, Prague.